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Effective Modeling Of Agricultural Practices Within Large-Scale Hydrologic And Water Quality Simulations

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EFFECTIVE MODELING OF AGRICULTURAL PRACTICES WITHIN LARGE-SCALE HYDROLOGIC AND WATER QUALITY SIMULATIONS

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The previously developed watershed hydrological and water quality model for St. Louis Bay watershed by Kieffer (2002) was refined and calibrated. The aspects of model development refinement included development of fertilization-related nutrient input parameters, evaluation of nutrient input methods, development of plant uptake-related nutrient input parameters, non-cropland simulation using PQUAL module, and recalibration of hydrology in Jourdan River. The related information of typical cropland management practice based on consultation from Mississippi State University Extension Service (MSU-ES) personnel was integrated into the watershed model. In addition, the Mississippi Department of Environmental Quality (MDEQ) observed water quality data were analyzed to evaluate the appropriateness of current watershed delineation and assess the health of the stream based on the MDEQ proposed numerical water quality target. The refined watershed model was calibrated in Wolf and Jourdan Rivers using both

United States Geological Survey (USGS) and MDEQ observed water quality data. The concentrations of water quality constituents calculated from the developed watershed model will be provided as boundary conditions for the developed Bay hydrodynamic and water quality model for Total Maximum Daily Load (TMDL) studies.

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CHAPTER I

INTRODUCTION

Among the assessed 19% of total miles of U.S.A's rivers and streams, 39% is impaired (EPA, 2000). The impairment of water quality is mainly caused by point and non-point source pollutants. The reduction of point source pollutants since the late 1960's has reached its practical threshold and still can not solve the water quality problem, hence, more attention has been focused on non-point source pollutant control (Hosseinipour and Heatwole, 1995; Sharpley and Rekolainen, 1997). The non-point source from agriculture has been identified as the leading cause of water quality impairment (EPA, 2000).

The Clean Water Act (CWA) was enacted in 1972 by the U.S. Congress, with the objective to restore and maintain the physical, chemical, and biological integrity of the nation's waters and ensure that all the United State's waters are suitable for their intended uses. To this end, the CWA has resulted in the concept of Total Maximum Daily Load (TMDL) for all the impaired water bodies listed in the 303(d) List. As defined by U.S. EPA, a TMDL is "a calculation of the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards, and an allocation of that amount to the pollutant's sources." The sources include point sources, non-point sources, background sources, and a margin of safety.

The point source pollutants are comparatively easy to be specified in terms of types, magnitude and locations; the point source data could be found in the National Pollutant Discharge Elimination System (NPDES). However, the non-point source pollutants are much more difficult to be characterized due to their diffusive nature in the watershed. Watershed modeling is a very effective approach to characterize the non-point source loadings from different land uses. In addition, watershed computational models are very useful analysis and planning tools to help identify primary watershed processes, quantify the contributions from different loading sources, guide further data collection, and evaluate the effects of Best Management Practices (BMP). Hence, watershed computational model is often chosen as a TMDL determination tool to devise the load allocation scheme.

St. Louis Bay, along with its two major tributaries, Wolf River and Jourdan River are included in the Mississippi 1998 Section 303(d) List for violation of the designated water use purpose of recreation and shellfish harvesting. In 1997, the modeling research of St. Louis Bay water quality was initiated by Mississippi State University to develop a loosely coupled modeling system for Mississippi Department of Environmental Quality (MDEQ) for the fecal coliform TMDL determination purpose (Hashim, 2001; Huddleston et al., 2003). In the above modeling efforts, the developed coupled modeling system included a watershed hydrology and water quality model and a bay hydrodynamic and water quality model. Environmental Fluids Dynamics Code (EFDC) was applied to create the water body modeling domain and simulate the hydrodynamics and fecal coliform transportation in the bay. The Better Assessment Science Integrating Point and Nonpoint Sources (BSINS2.0) - Nonpoint Source Model (NPSM) was selected to be the watershed model, which calculated the flow and fecal coliform loadings from the Wolf River and Jourdan River watersheds to the bay.

Since nutrients and Dissolved Oxygen (DO) are considered to be very important of healthy indexes of St. Louis Bay aquatic ecosystem, the modeling efforts was extended to include DO and nutrients (Kieffer, 2002). The previous watershed model, NPSM, was converted to Hydrological Simulation Program Fortune (HSPF) to keep the watershed hydrology model unaltered. The modeling performance of DO and Biochemical Oxygen Demand (BOD) was reasonable. However, the simulated nutrients including NO_3, NH_3, and PO_4, were one or two orders higher than the observed data.

Extensive efforts have been spent on reviewing the applications of HSPF, as shown in the next chapter. It was found that much more efforts have been spent on how to calibrate the model instead of how to develop the loading forcing function and how to correctly input the developed function into the model. How to develop a representative linkage between pollutant loading sources and in-stream concentrations is the core part of watershed water quality modeling. As indicated by Chapra (2003), without correct estimation of boundary loading functions, the model calibration would become a meaningless exercise. In addition, it is very important to make sure that the model generates your intended boundary loadings. Different input methods have different interpolation functions; hence for the same loading function, different input methods will generate different pollutant loadings. Further, some input parameters of boundary loading function are model-specific. These parameters have to be calibrated to their intended values; otherwise, the watershed water quality processes will be misrepresented.

The large discrepancies between simulated and observed nutrient concentrations by the developed St. Louis Bay watershed model indicated that there was something wrong with the boundary loading functions. The general objective of this research is to assess HSPF in simulating water quality constituents, especially the AGCHEM and PQUAL module. The specific objectives include:

- Developing the fertilization-related input parameters for long-term simulation.
- Evaluating influence of cropland fertilization practices on nutrient input parameters for watershed modeling.
- Evaluating different nutrient input methods for AGCHEM and their impacts on modeling performance.
- Developing and determining the model-specific input parameter of plant uptake for corn, hay, soybean and wheat.
- Evaluating the impacts of plant uptake forms of nitrogen (ratio of nitrate to ammonia) on plant uptake and nitrogen outflow from cropland.
- Developing and calibrating the model-specific input parameter of accumulation rate of pollutants for non-cropland.

Sometimes, the accumulation of modeling experience is a painful process; you may find that the calibrated model half a year ago was completely wrong due to the wrong input unit or some other mistakes. But only through these painful learning processes can modelers gain more experience. I would like to finish this chapter with this sentence; "…modeling is a process, not an end (Chapra, 2003)."

CHAPTER II

LITERATURE REVIEW

The literature review will only focus on the historical development of HSPF, data requirement, and its applications. The detailed documentations of model structure and calculation algorithm were described by Hashim (2001), Huddleston et al. (2001), Kieffer (2002), and Huddleston et al. (2003).

Development History of HSPF

HSPF is a comprehensive watershed hydrology and water quality model that allows the integrated simulation of soil pollutant transportation with in-stream hydrodynamics and water quality processes. HSPF has experienced more than 40 years development, testing and refinement since 1960s (Fig. 2.1-1). The watershed hydrology model of HSPF originated from the Stanford Watershed Model (SWM) developed by Crawford and Linsley (1966). The SWM was further refined and resulted in the creation of Hydrocomp Simulation Program (HSP) (Hydrocomp, 1969), which allowed for the non-point source simulation. These two models constructed the theoretical basis of hydrologic and hydraulic simulation of HSPF.

In 1970s, the development of pollutant transportation models contributed to the continual refinement of water quality simulation of HSP. The Pesticide Transportation and Runoff (PTR) model was developed by loosely coupling the applied pesticide onto the water and sediment movement simulated by SWM (Crawford and Donigian, 1973). Further modification and refinement of PTR resulted in the development of Agricultural Runoff Management (ARM), which included soil nutrient transformation simulation (Donigian and Crawford, 1976a). The ARM was further improved by Donigian et al. (1977) by including plant nutrient uptake and other refinement. The Nonpoint Source (NPS) model was developed to meet the need of assessing the pollutant sources in major metropolitan areas (Donigian and Crawford, 1976b).

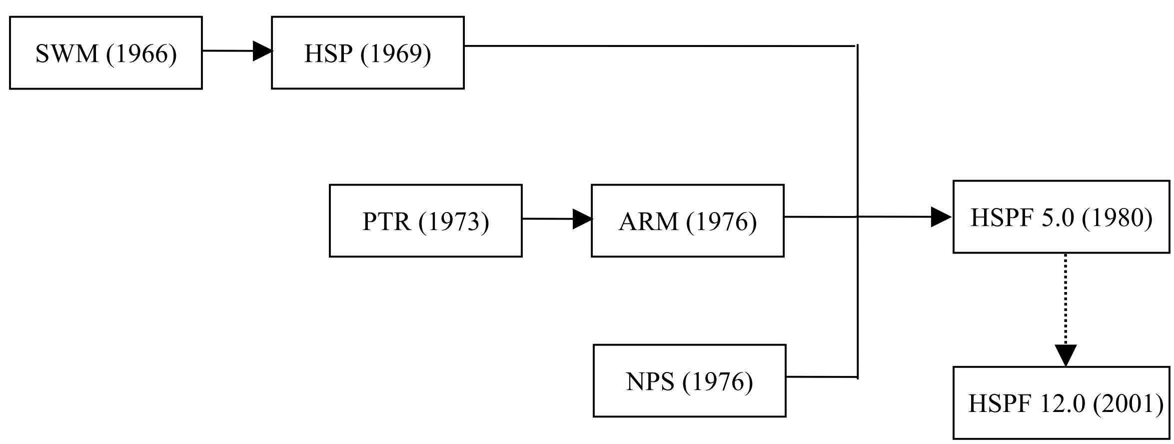


Fig.2.1- 1 Development history of HSPF

The combination of all essential functions of HSP, ARM and NPS resulted in the creation of HSPF 5.0 (Johanson, 1980). Since then, HSPF has experienced continual modification and refinement and the latest version is HSPF 12.0 (Bicknell et al, 2001).

Data Requirement of HSPF

The application of HSPF requires extensive datasets. The data required to simulate watershed hydrology include spatial, meteorological, and monitored flow data. Among the spatial data, DEM and stream network data are used to estimate contributing area and delineate sub-watersheds; landuse data are used to determine the area and relative position of different land use categories; State Soil Geographic (STATSGO) database is used to determine soil texture and estimate the related hydrological input parameters. The computational interval of HPSF is hourly; hence, it requires hourly input of meteorological data. The required meteorological data are precipitation, air temperature, dewpoint temperature, wind movement, solar radiation, cloud cover, potential evapotranspiration and surface evaporation. The precipitation drives the watershed hydrology modeling. The accuracy of the input precipitation data determines the reliability of developed model. The best case is that there is a meteorological station for each sub-watershed to capture the spatial variation of rainfall over the entire watershed. When hourly data are not available, Watershed Data Management Utility (WDMUtil) software can be used to disaggregate the daily data based on the hourly precipitation pattern in adjacent stations. The observed flow data are used to calibrate and evaluate the developed model.

For water quality constituent modeling, more data have to be provided in addition to the above required for hydrological modeling. For example, to model the dynamic transportation process of nutrient using AGCHEM module, the following data need to be provided: observed nutrient in-stream concentrations, tillage practice, cropland-specific fertilization practice including fertilization method and timing, contribution from manure application, the amount and timing of plant uptake, the first order rate of transformation processes such as mineralization, immobilization, nitrification, denitrificaiton, and sorption/adsorption, and so on.

HSPF Applications

The applications of HSPF could be found in the journal papers, conference papers, edited books, reports, and internet. Journal papers are easier to locate for the academic community than other media. In addition, theoretically, the results of peer-reviewed papers are more reliable than in other media. A total of 43 applications of HSPF from 1980s to 2005 were found in 17 different academic journals, and these applications were reviewed and summarized (Table A-1).

HSPF has been successfully applied in different geographical regions including glaciated watersheds, arid watersheds, agricultural watersheds, urban watersheds, and undeveloped watersheds. The applications of HSPF focused on the following areas (Table A-1):

- Assess water quality and quantity in a watershed
- Evaluate non-point source pollution from agriculture
- Evaluate the effects of best management practices (BMPs)
- Assess the impacts of urban development on watershed hydrology and water quality
- Develop techniques to help model calibration

- Enhance modeling performance of in-stream temperature by integrating with other computer program
- Evaluate modeling performance of HSPF in different geographic regions.
- Assess the usefulness of BASINS database
- Compare the modeling performances between HSPF and other models
- Evaluate the effects of global climate change on watershed hydrology
- Assess the sensitivity of input parameters
- Integrate HSPF with MODFLOW to estimate the total water balance
- Evaluate impacts of fertilization practices on in-stream nutrient simulation
- Test the applicability of HSPF by using meteorological data from global circulation model (GCM)
- Evaluate the sensitivity of HSPF hydrograph to three land cover map inputs

The applications of HSPF were propelled by the enhancement of computer capacity and popularity of desktops; only 3 applications were published in 1980s, 10 in 1990s, and 30 from 1991 to present (Table A-1). This trend of HSPF publications coincides with the development of computer capacity and popularity. It can be anticipated that more papers would be published in the future since HSPF can be conveniently run on the desktops in the office.

HSPF is able to simulate streamflow, sediment, nutrient, fecal coliform, pesticide, and conservative substance. Of the total 43 applications, the most frequently modeled constituent was still streamflow alone (Fig. 2.3-1). This trend did not change even in the 2000s; of the total 30 applications of HSPF, 18 were only confined to hydrological modeling. The reason why so many applications focus on hydrology-related topics is that a sound calibrated hydrology model is the prerequisite of further water quality modeling. In addition, the extensive data requirement of HSPF may limit its applications to water quality constituents. Further, the complicated module structures, and lack of documentations in development of water quality input parameters and technical operation created problems for users and confined its application. Finally, there is lack of modeling guideline for water quality modeling. The modeling guideline of hydrological modeling for HSPF, proposed by EPA (2000), outlined the detailed steps to facilitate the modeling users to calibrate the model. A guideline for water quality modeling, even a very rough one, would be a very useful tool to help facilitate the users.

HSPF is very flexible to simulate both small and large watersheds, with modeled watershed areas ranging from 0.07 to 386,102.16 mile^2 (Table A-1 and Fig. 2.3-2). Novotny and Chesters (1981) pointed out the negative linear relationship between the model reliability and watershed size for different constituents (Fig. 2.3-3). For a large watershed, it is very difficult to correctly characterize the watershed hydrological and water quality processes. Generally, HSPF was seldom applied to watershed larger than 5,000 mile^2 (Fig. 2.3-2).

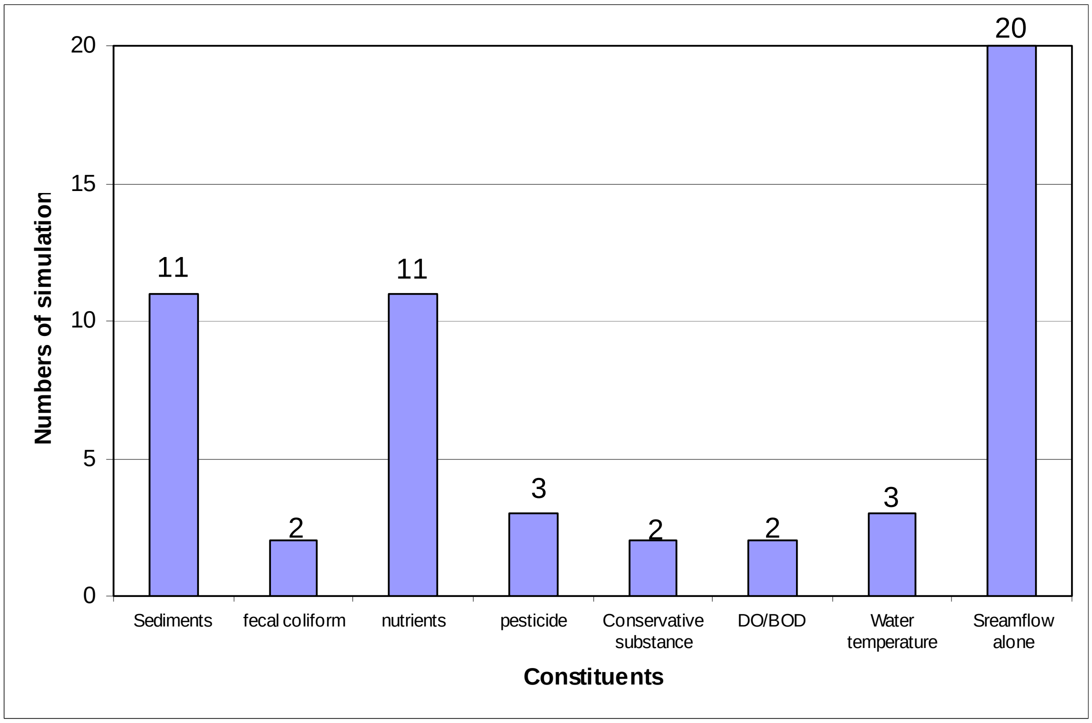


Fig.2.3- 1. Simulated constituents by HSPF (Table 2.3-1).

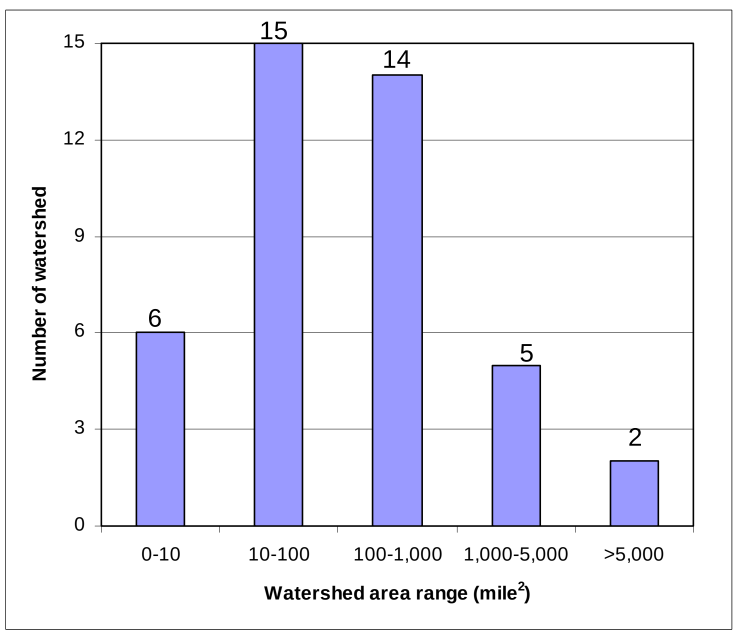


Fig.2.3- 2. Watershed area distribution of HSPF applications (Table 2.3-1).

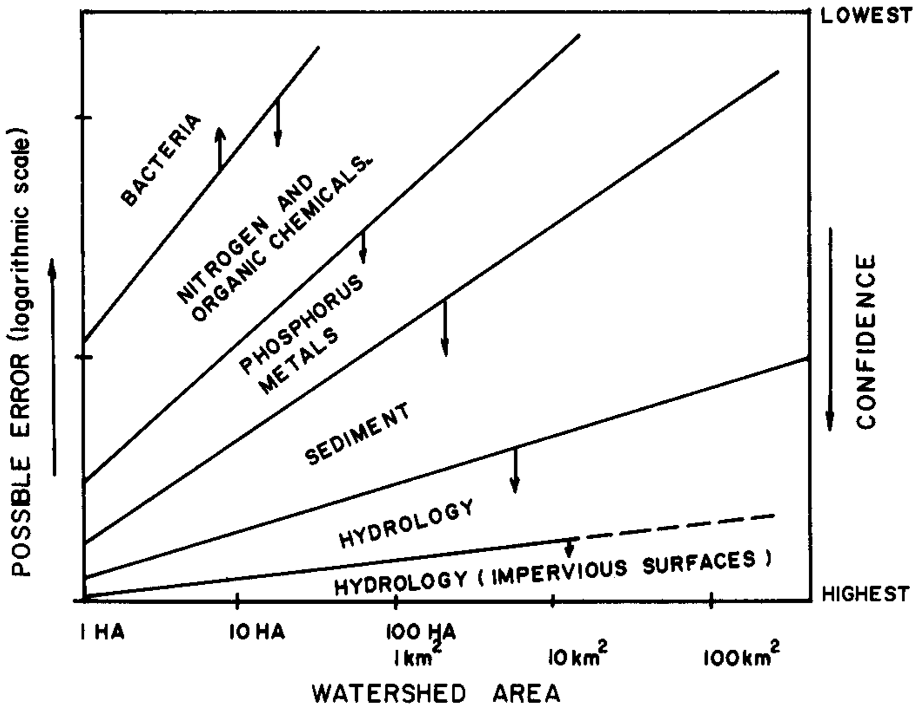


Fig.2.3- 3. Relationship between the model reliability and watershed size for different constituents (After Novotny and Chesters, 1981)

Of the 43 applications of HSPF, the duration was not specified in three applications. HSPF is not only able to simulate short-term storm events but also to model long-term hydrological and water quality processes. The duration of simulation period ranged from 1 month to 40 years (Fig.2.3-4 and Table A-1).

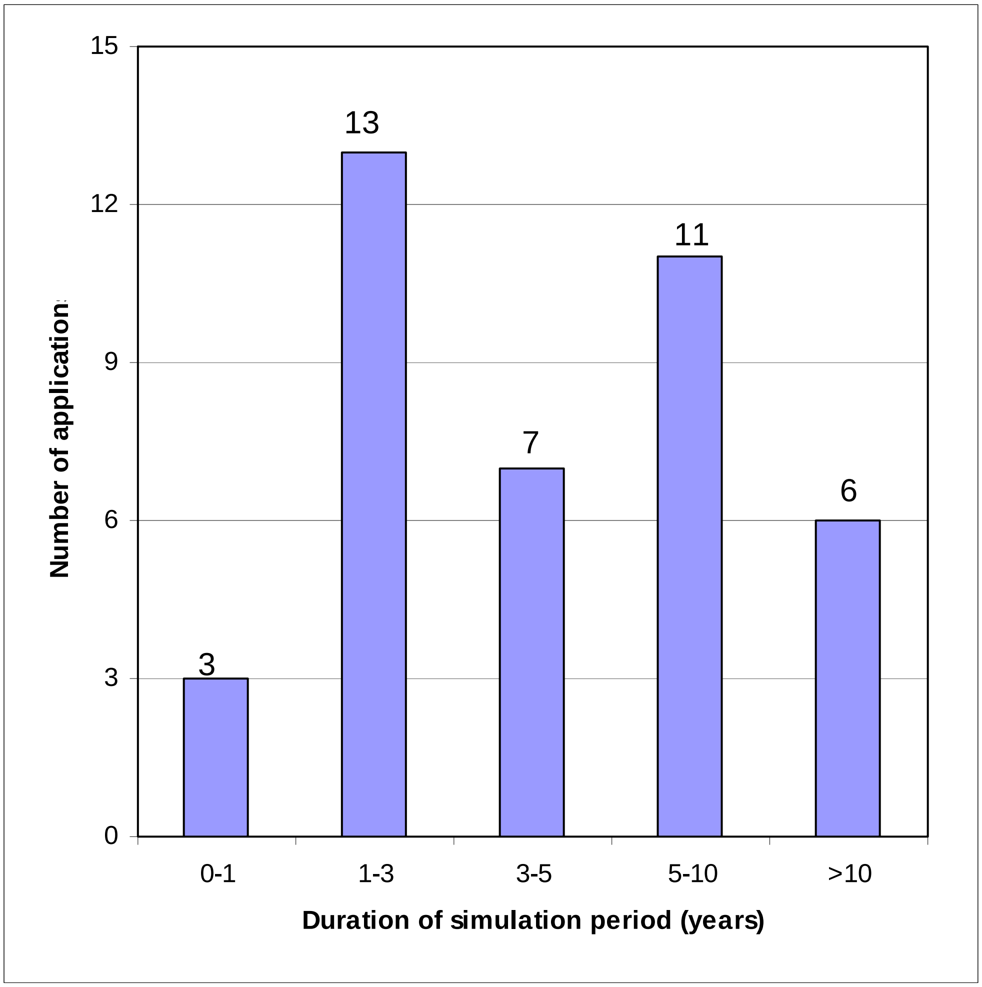


Fig.2.3- 4. Simulation period duration of HSPF applications (Table 2.3-1).

The modeling purpose of HSPF applications is to develop a watershed model with predictive ability. A watershed model with predictive ability should be calibrated under different climate, soil moisture and water quality conditions. The recommended simulation duration was at least 3 to 5 years (Donigian, 1999). For applications with simulation duration less than three years, it had better include both dry and wet years; otherwise, the prediction ability of the developed model is questionable. As long as there is no obvious change of land use, long-term simulation of HSPF would give more confidence in model predictive ability. However, the required dataset to support developing the input parameters for long-term simulation would be much more difficult to obtain than short-term simulation.

The comparison studies between HSPF and SWAT indicated that the application of HSPF requires comparatively more time and effort to prepare dataset and calibrate the model (Van Liew et al., 2003; Borah and Bera, 2004; Saleh and Du, 2004; Singh, et al., 2005). HSPF requires more input parameters to be developed and estimated than SWAT, and hence is deemed to be less user-friendly (Singh, et al., 2005). SWAT performed better in predicting the low or extreme low flow events than HSPF (Van Liew, et al., 2003; Singh, et al., 2005). The requirement of human resources of HPSF is extensive (Borah and Bera, 2004). It is a very difficult and time consuming task to conduct comparison studies between HSPF and SWAT since both models needs extensive dataset and modeling expertise. Hence, it is dangerous to make a hasty statement regarding which model is better based on the simple numerical evaluation criterion, such as determination coefficient (R^2). The low value of determination coefficient could be associated with data limitation or incorrect characterization of loading sources. Even some model deficiencies could be overcame by using advanced manipulations.

CHAPTER III

ANALYSES OF STREAM DATA AND HEALTH OF STREAM

The observed water quality data, especially the high-quality data, are very important to calibrate and evaluate the performance of the developed model. The analysis and assessment of water quality data can help identify the major environmental problem in the study area and determine the modeling purpose. The extent of spatial variations of water quality parameters may give insight into how may sub-watersheds the modeler should delineate; more sub-watersheds need to be divided in order to capture the higher spatial variations of pollutants of concern. The objective is to analyze the MDEQ observed water quality data to better understand the aquatic ecosystem, capture the spatial variation of water quality parameters, and evaluate the appropriateness of the delineation of the developed St. Louis Bay watershed model based on the analysis of the observed data. Cluster analysis was also used to classify the sampling stations. Finally, the representation of spatial distribution of landuse in the delineated sub-watershed was presented.

Data Description

The majority of highest quality data available to calibrate the developed water quality model of the Bay St. Louis watershed were collected by MDEQ (2002). A total of

16 stations were selected by MDEQ to monitor the physical and water quality parameters, and the locations of the sampling stations are shown in Fig. 3.1-1.

The water quality data were collected from the sampling stations during 4 baseflow events and 8 storm events from water year 2000 to 2001. The sampling wet weather events were distributed throughout the year to capture the seasonal variation in water quality. The sampling of storm events occurred following a dry inter-event period of at least 72 hours, whereas the sampling of baseflow events was conducted at least 72 hours after the last storm events.

Assessment of Water Quality Parameters

The observed water quality parameters include DO, TSS, COD, BOD, TOC, TP, PO_4, TKN, NH_3, NO_3, fecal coliform, and chlorophyll a. The analysis was confined to DO, BOD, TP, PO_4, TKN, NH_4, NO_3, and chlorophyll a. MDEQ (2000) proposed the "target level" for some water quality parameters having no specified numerical criteria (Table 3.2-1). The target levels are developed based on best professional judgments and literature review. The observed water quality data were compared with these target levels as indicators of potential water quality problems. The monitoring data from stations WR2, WR3, WR4, WR5, CRN1, BP3, and JNB2 were analyzed to assess the water quality condition. WR2, WR3, WR4, WR5, and CRN1 are located in the Wolf River system, whereas BP3 and JNB2 are located in the bayou areas. Hence, this analysis allows us to understand the water quality conditions in both the Wolf River system and bayou areas. The data from station WR5 were excluded from the analysis due to the small sample size.

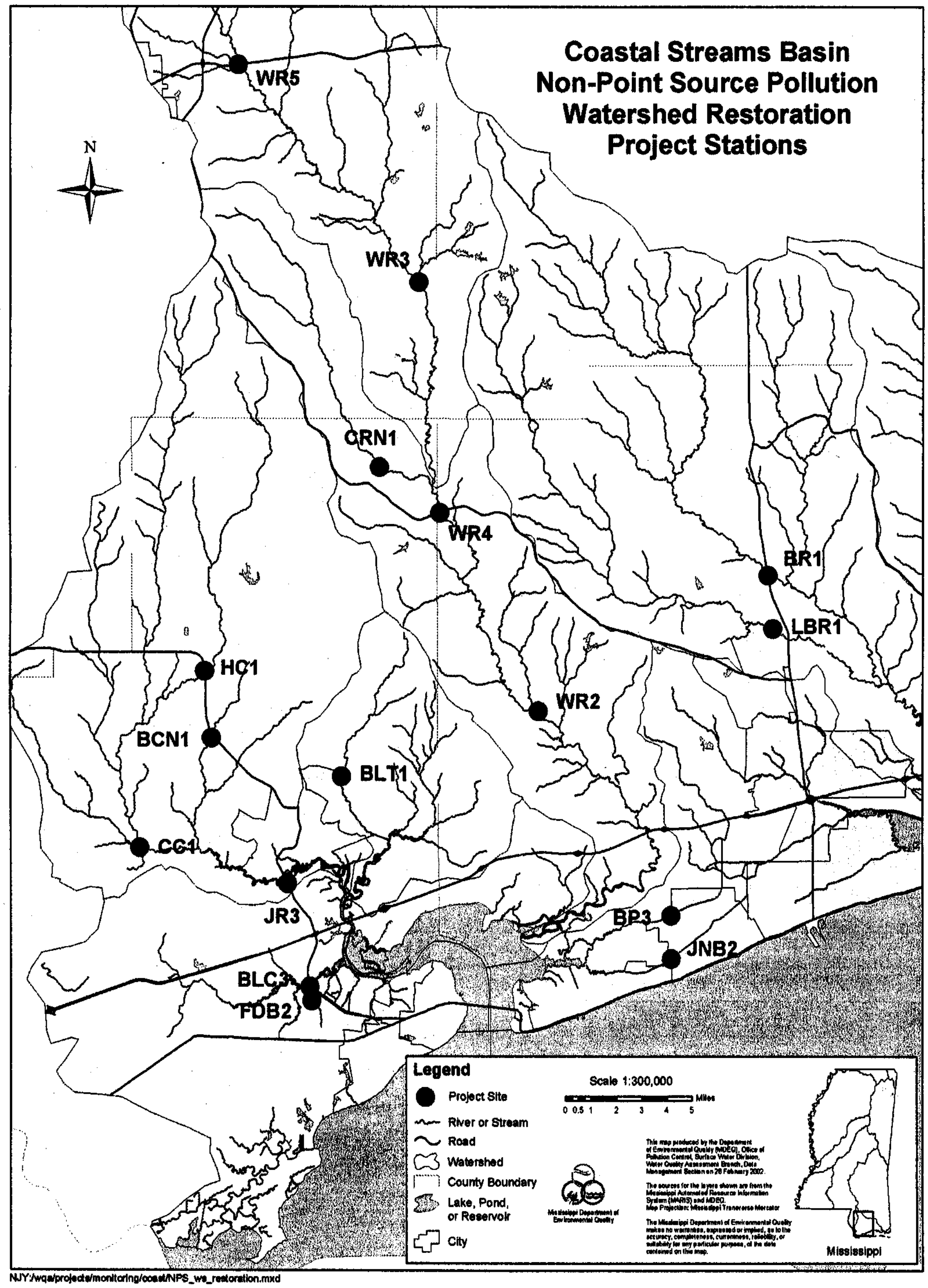


Fig.3.1- 1. Sampling stations in the Bay St. Louis watershed (After MDEQ, 2002).

Table 3.2- 1. Water quality target level proposed by MDEQ (2000)

WATER QUALITY PARAMETER	UNITS	TARGET LEVEL
Dissolved Oxygen (DO)	mg/L	> 4.0
Biochemical Oxygen Demand (BOD)	mg/L	< 5
Nitrate and Nitrite	mg/L as N	< 1
Ammonia	mg/L as N	< 1.3
Total Phosphorus	mg/L as P	< 0.2
Chlorophyll a	Mg/L	<0.01

Assessment of DO

For the Wolf River system including stations WR2, WR3, WR4, and CRN1, DO was at good conditions with average concentration of 9.29 mg/L and minimum value of 5.60 mg/L, higher than 4.0 mg/L, the proposed target by MDEQ (Table 3.2-1). However, for the stations BP3 and JNB2, some low DO events occurred; BP3 had the minimum DO of 2.58 mg/L and JNB2 had the minimum value of 2.94 mg/L (Fig. 3.2-1). The reasons leading to these low flow events in the bayou stations could be attributed to several factors. First, the topography in the bayous is very flat and hence, the water in the bayous does not circulate very well, which would result in a lower value of aeration coefficient of DO. In addition, there is more human influence in bayou areas than in the Wolf River; the majority of the urban area concentrates near the coastal area. The wastewater discharge from urban area could consume DO and result in low DO events.

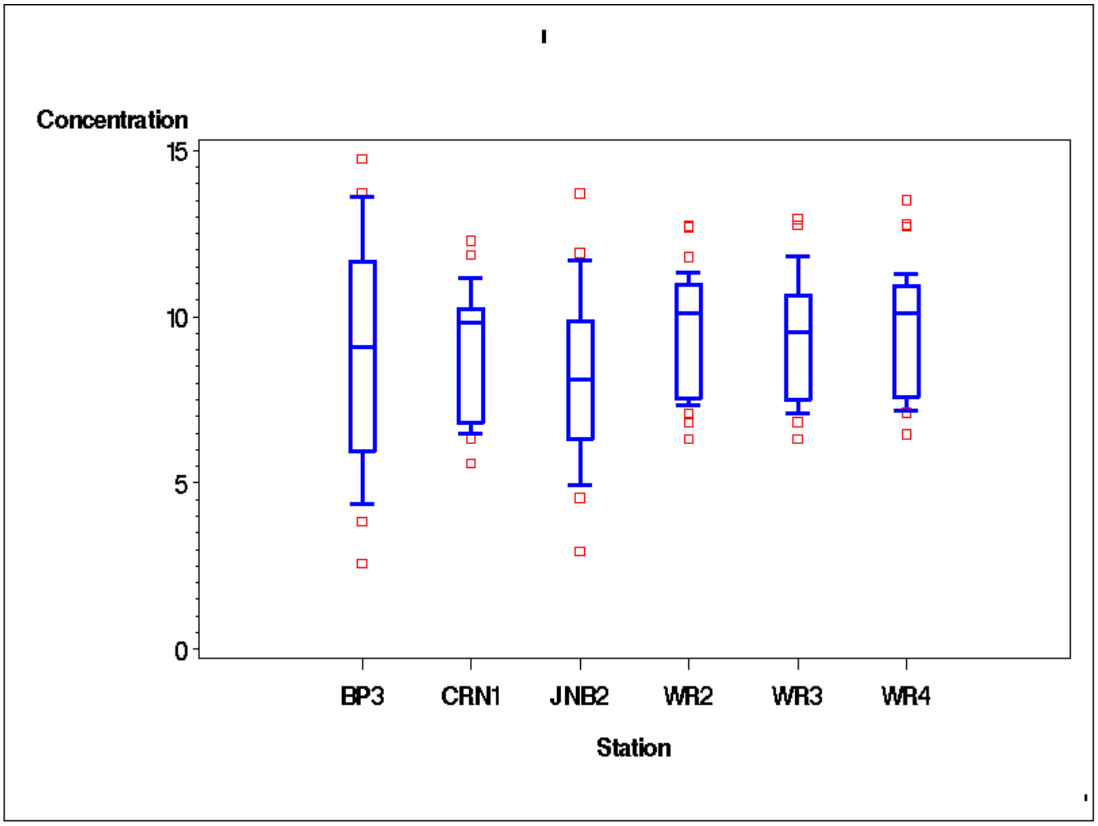


Fig.3.2- 1. Boxplots of DO in the sampling stations.

The results of linear regression analysis indicated that there was strong negative linear relationship between DO concentrations and water temperature in the Wolf River (Fig. 3.2-2). The coefficient of determination (r^2) is 0.8243, and the linear model could be used to predict the DO concentrations using water temperature. However, the linear relationship was weak in the bayou areas with r^2 of 0.2899 (Fig. 3.2-3). The low value of r^2 is because the linear model is not able to account for the anthropogenic effects and alteration of flow regime in the bayou area.

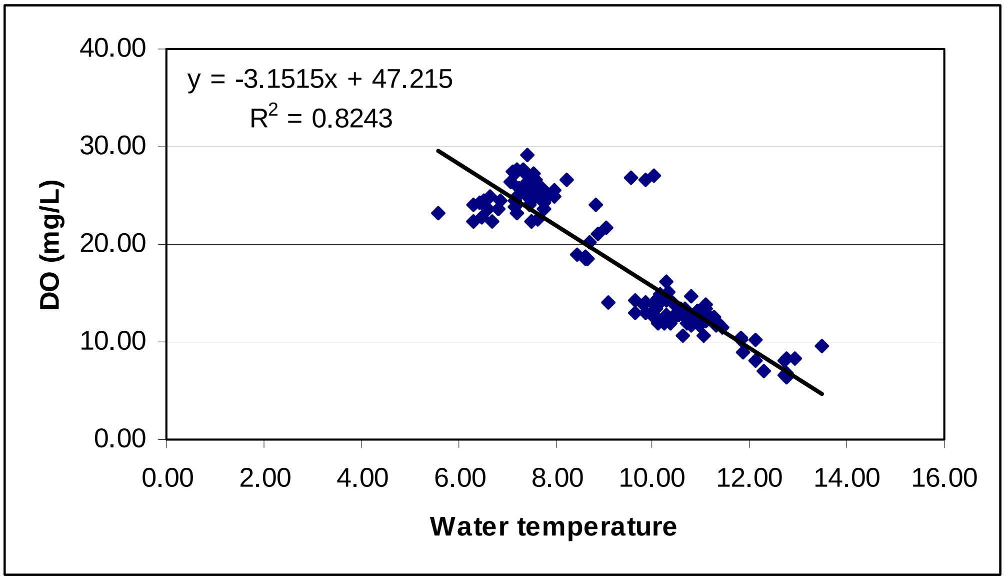


Fig.3.2- 2. Linear regression analysis between DO and water temperature for the Wolf River system including stations WR2, WR3, WR4, WR5, and CRN1.

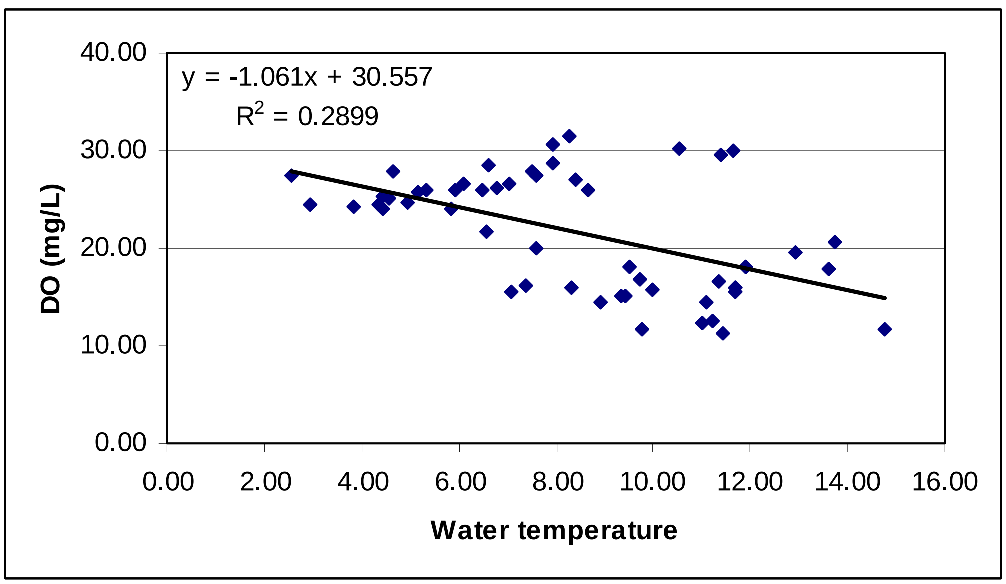


Fig.3.2- 3. Linear regression analysis between DO and water temperature for the bayou areas including BP3 and JNB2.

The temporal distributions of DO trend were very consistent at the sampling stations located in the Wolf River system, but less consistent for the stations in the bayou area (Fig. 3.2-4 and 3.2-5). For a specific sampling time, the variation of observed DO concentrations among the stations in the Wolf River was very small (Fig. 3.2-4), but very large in the bayou areas (Fig. 3.2-5). The extent of consistency in DO trends between two sampling stations can be reflected by the correlation analysis; higher values of r^2 indicate more consistent trends. All the coefficients of determination among the stations in the Wolf River system were higher than 0.90 indicating a very consistent trend in temporal DO distribution (Table 3.2-2). The coefficients of determination between stations in the bayou areas (stations BP3 and JNB2) and any other stations were less than 0.75, indicating a less consistent trend of DO distribution (Table 3.2-2).

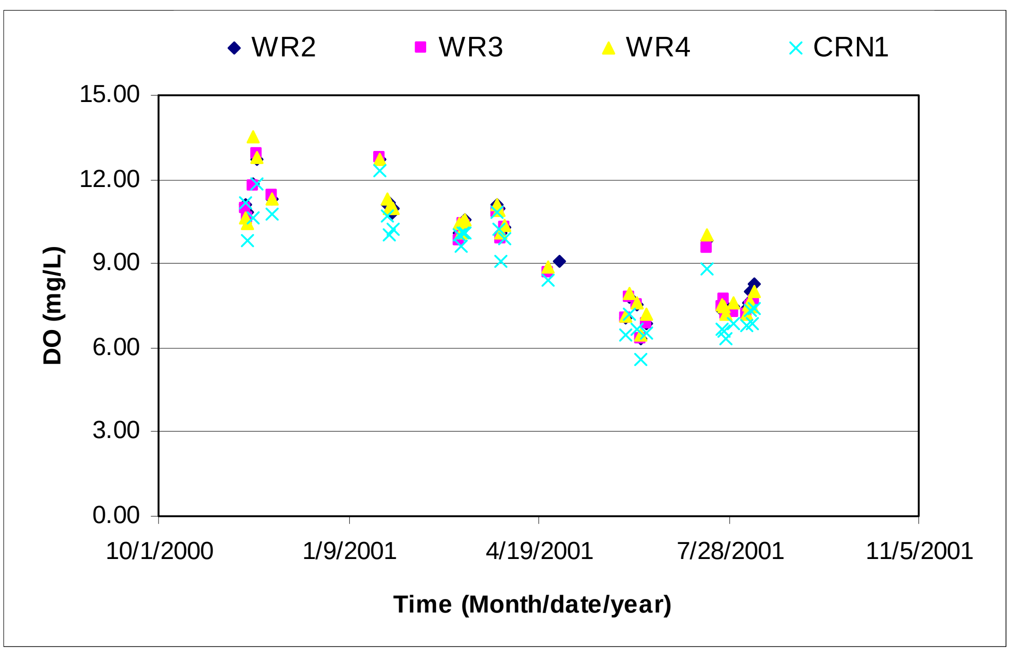


Fig.3.2- 4. Consistent temporal distribution of DO trend in the Wolf River system.

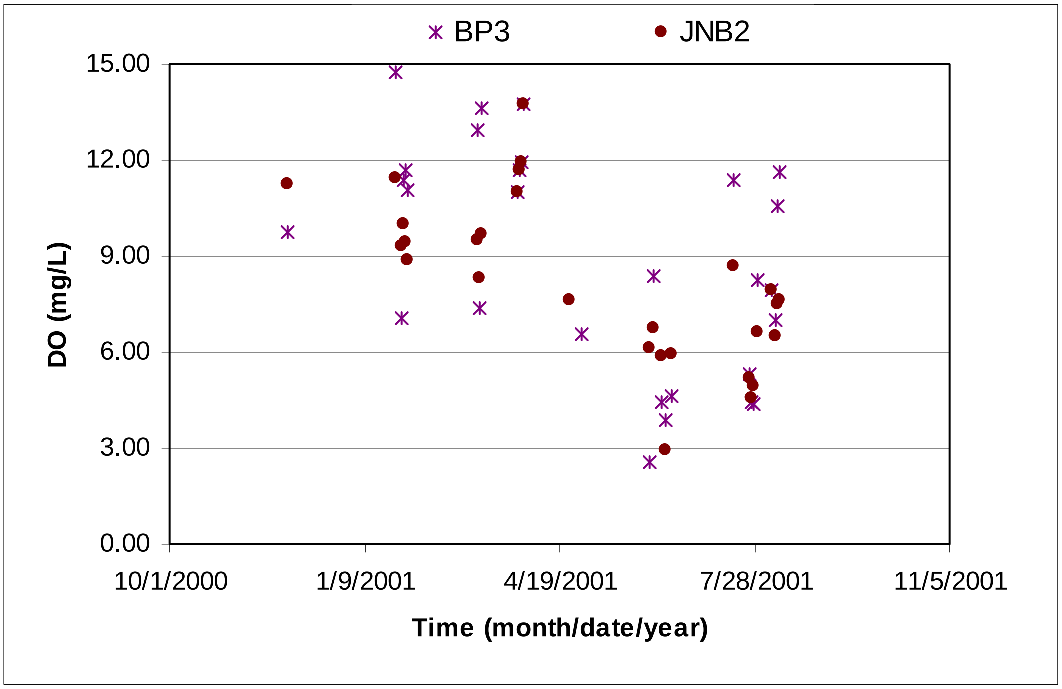


Fig.3.2- 5. Inconsistent temporal distribution of DO trend in the bayous.

Table 3.2- 2. Coefficients of determination of DO among stations.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.9903	0.9672	0.9749	0.673	0.7313
WR3	0.9903	1	0.9667	0.9663	0.6391	0.7317
WR4	0.9672	0.9667	1	0.9335	0.6704	0.7316
CRN1	0.9749	0.9663	0.9335	1	0.6866	0.7464
BP3	0.673	0.6391	0.6704	0.6866	1	0.6972
JNB2	0.7313	0.7317	0.7316	0.7464	0.6972	1

Assessment of BOD

In general, BOD in the surface water of Wolf River was at a low level; the average BOD concentrations at all the sampling stations were less than 5.0 mg/L, the proposed target by MDEQ (Fig. 3.2-6 and Table 3.2-1). The frequency distribution of

BOD concentrations of all the samples was displayed in Fig. 3.2-7. The majority of the samples had very low BOD concentrations; BOD concentrations in 83% of the total samples were less than 3.0 mg/L. Only 6% of the samples had BOD concentration higher than 15 mg/L (Fig. 3.2-7) and the highest BOD concentration was 34.4 mg/L.

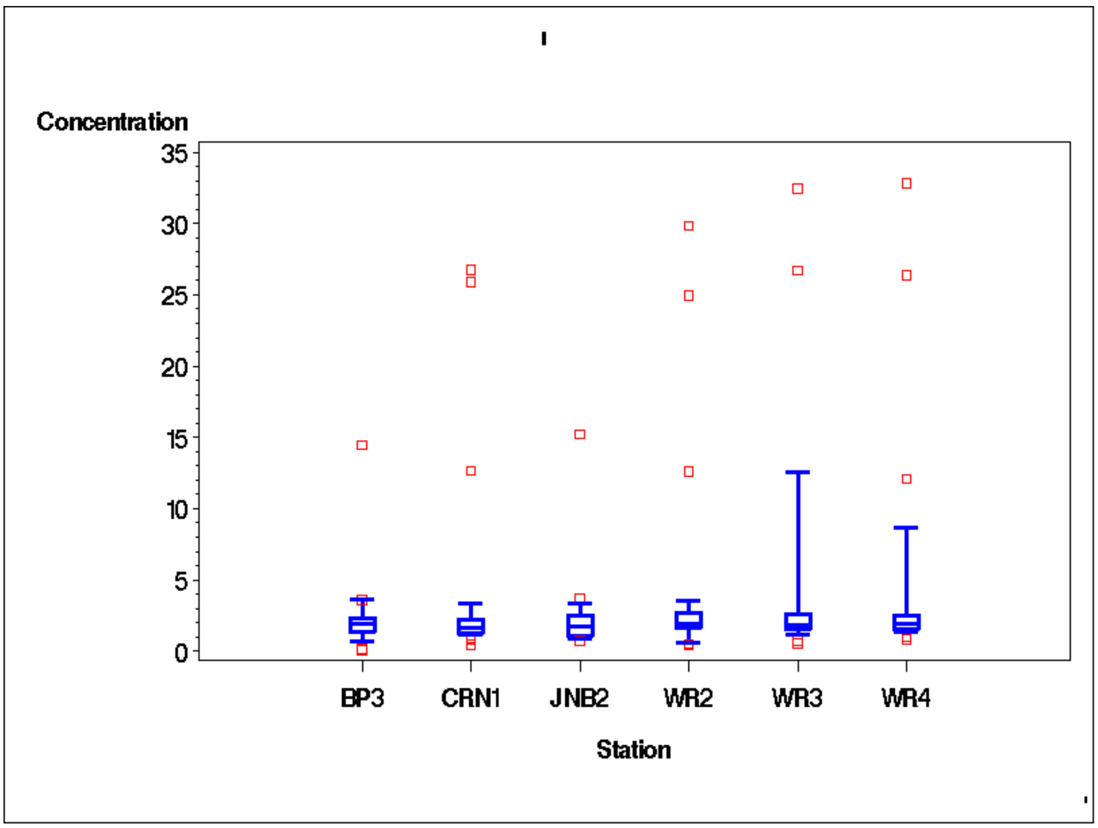


Fig.3.2- 6. Mean concentration of BOD in the sampling stations.

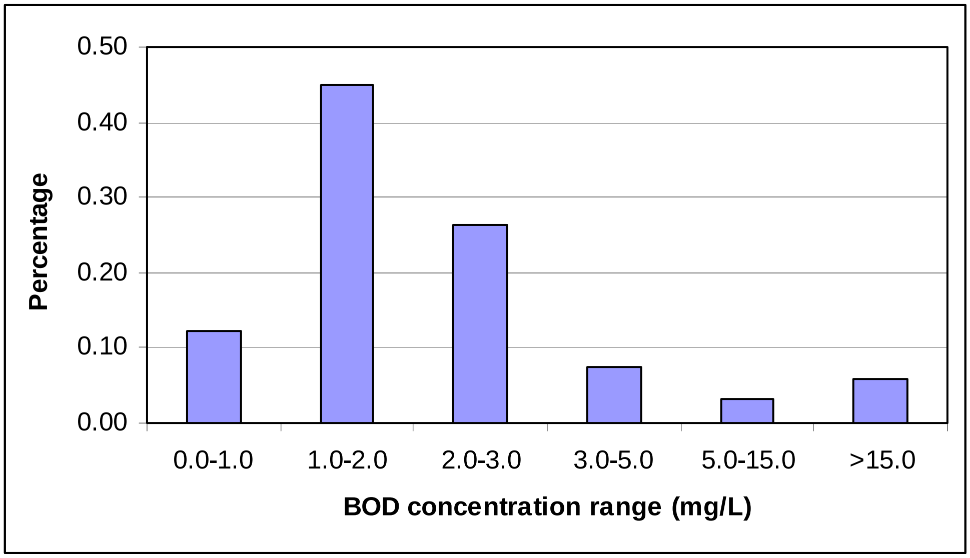


Fig.3.2- 7. Frequency of BOD concentration distribution in all samplings.

Different from the temporal distribution of DO concentrations, the temporal distribution of BOD was consistent at all the sampling stations. The determination coefficients of BOD among all the stations were all higher than 0.89, indicating a strongly consistent temporal trend at the sampling stations (Table 3.2-3).

Table 3.2- 3. Coefficients of determination of BOD among stations.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.99959	0.9794	0.9907	0.9092	0.899
WR3	0.9959	1	0.9814	0.9882	0.9515	0.9184
WR4	0.9794	0.9814	1	0.9668	0.9419	0.9056
CRN1	0.9907	0.9882	0.9668	1	0.9252	0.891
BP3	0.9092	0.9515	0.9419	0.9252	1	0.9642
JNB2	0.899	0.9184	0.9056	0.891	0.9642	1

Assessment of TP and PO_4

The majority of the samples had high concentrations of TP; TP concentrations in 91% of the total samples were higher than 0.2 mg/L, the recommended threshold target by MDEQ (Fig. 3.2-8 and Table 3.2-1). This indicated that phosphorous could be a potential water quality problem for the Wolf River watershed.

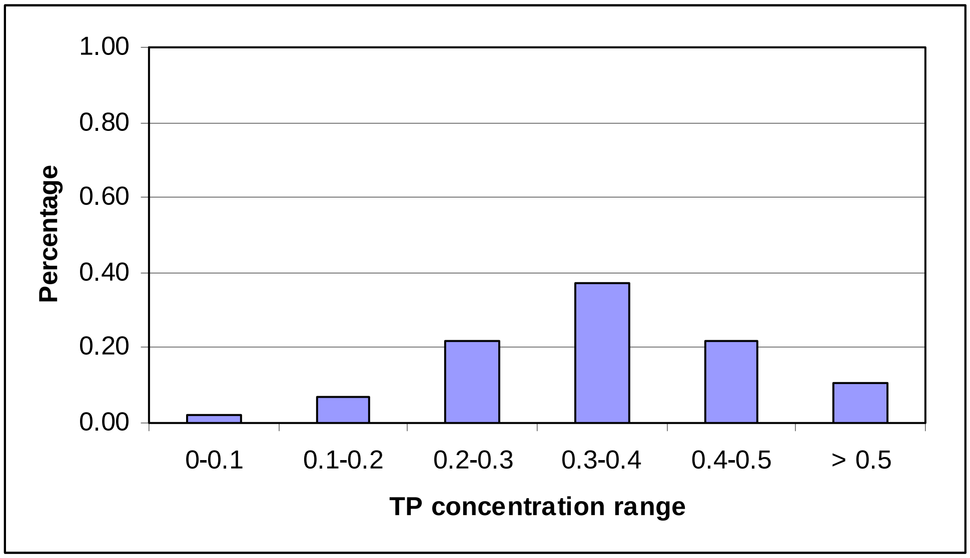


Fig.3.2- 8. Frequency of TP concentration distribution in all samplings.

The calculated mean values of PO_4 concentrations at all the sampling stations were higher than 0.2 mg/L (Fig. 3.2-9). Mean values of PO_4 concentration higher than 0.3 mg/L were observed in the station BP3 and JNB2. However, there is no recommended target value for PO_4 by MDEQ.

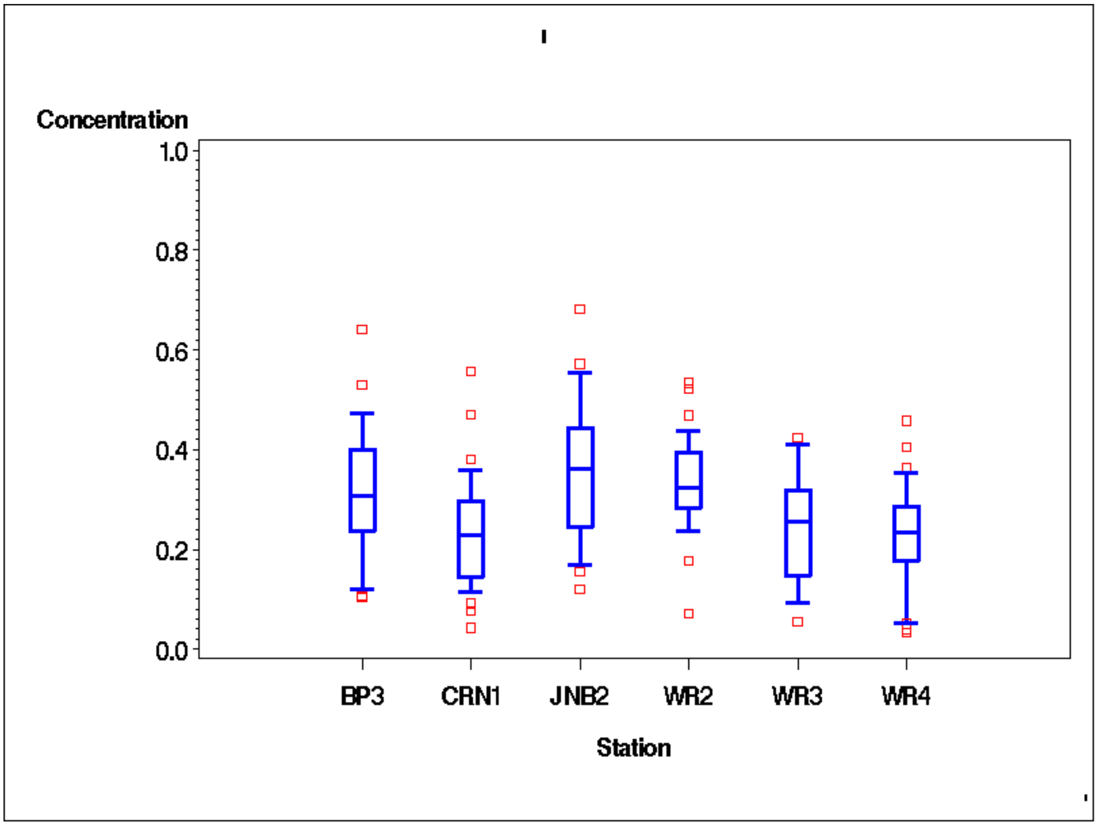


Fig.3.2- 9. Boxplots of PO_4 at the sampling stations.

Unlike DO and BOD, the temporal distribution of PO_4 is not consistent at all the sampling stations; most of the coefficients of PO_4 between the stations were lower than 0.5 (Table 3.2-4), indicating an inconsistent temporal trend. Only the coefficient of determination between WR3 and WR4 was higher than 0.5 (Table 3.2-4).

Table 3.2- 4. Coefficients of determination of PO_4 among stations.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.0375	0.0273	0.0246	0.1224	0.0386
WR3	0.0375	1	0.6348	0.463	0.4334	0.0131
WR4	0.0273	0.6348	1	0.389	0.096	0.0289
CRN1	0.0246	0.463	0.389	1	0.2719	0.0162
BP3	0.1224	0.4334	0.096	0.2719	1	0.1481
JNB2	0.386	0.0131	0.0289	0.0162	0.1481	1

Assessment of Total NO_3-NO_2 and NH_3

The NO_3 and NO_2 were measured together by MDEQ. In most cases, the major component of the total NO_3-NO_2 is in the form of NO_3 (Vousta *et al*. 2001). Hence, for the purpose of simplification, it was assumed that all the NO_3-NO_2 measured was in the form of NO_3. Median values of NO_3 concentrations at all the sampling stations were lower than 0.1 mg/L (Fig. 3.2-10), much lower than the MDEQ recommended target value for NO_3, 1.0 mg/L (Table 3.2-1). It was reported that the median value 0.1 mg/L of NO_3 concentration was found in unpolluted rivers (Meybeck, 1998). The maximum value of NO_3 concentration was 1.036 mg/L, much lower than 10.0 mg/L, the maximum permissible concentration for drinking water (Vousta *et al*. 2001). Median values of NH_4 at all the sampling stations were lower than 0.02 mg/L (Fig. 3.2-11), much lower than the MDEQ recommended target value for NH_4, 1.3 mg/L (Table 3.2-1).

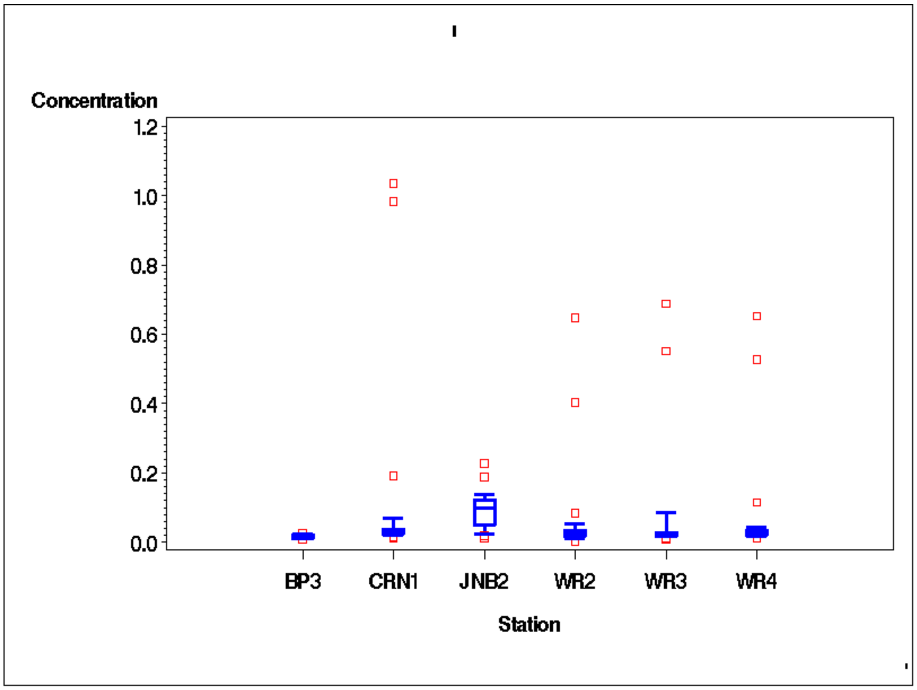


Fig.3.2- 10. Boxplots of the NO_3 at the sampling stations.

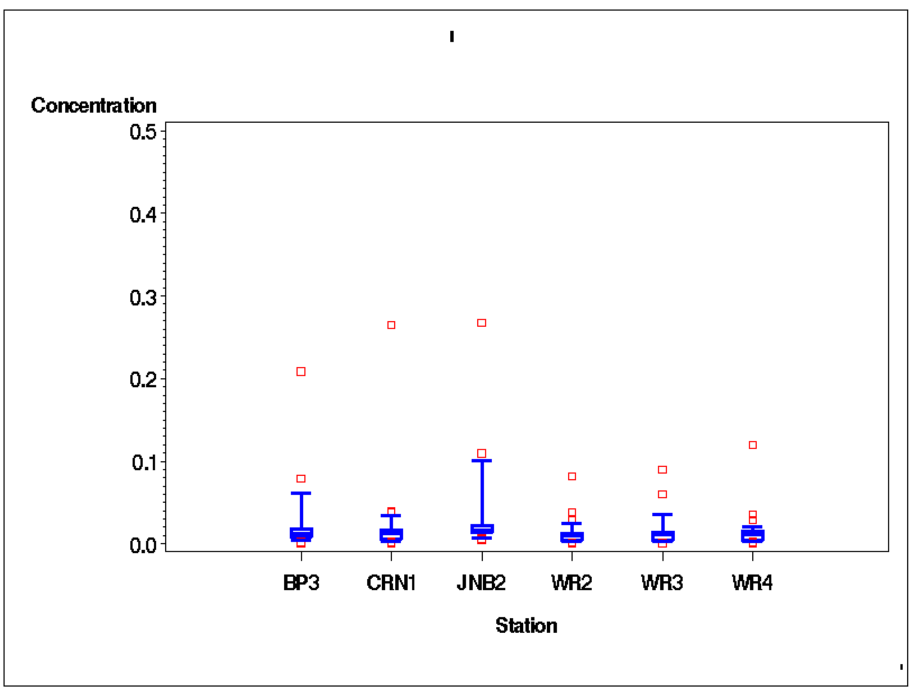


Fig.3.2- 11. Boxplots of NH_3 at the sampling stations.

Like DO, the temporal distributions of NO_3 concentrations were consistent for the stations in the Wolf River system; the values of determination coefficients were all higher than 0.85 (Table 3.2-5).

Table 3.2- 5. Coefficients of determination of NO_3 among stations.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.8677	0.8731	0.9473	0.2812	0.0006
WR3	0.8677	1	0.9958	0.9731	0.0013	0.0515
WR4	0.8731	0.9958	1	0.978	0.0452	0.1056
CRN1	0.9473	0.9731	0.978	1	0.0034	0.0402
BP3	0.2812	0.0013	0.0452	0.0034	1	0.02
JNB2	0.0006	0.0515	0.1056	0.0402	0.02	1

The determination coefficients of NH_3 among the sampling stations were shown in Table 3.2-6. The values of r^2 among WR2, WR4, CRN1, BP3, and JNB2 were all higher than 0.70, indicating a fairly strong consistent trend. Only WR3 had low value of r^2 with other stations. After carefully examining the original data, it was found that the measured NH_4 on Aug. 9, 2001 at WR3 was very low, whereas the NH_4 concentrations in other stations were comparatively high. The correlation analysis was re-conducted by removing the sample data on Aug. 9, 2001. The values of r^2 between WR3 and other stations were greatly increased (Table 3.2-7). From the view of statistics, the sample on Aug. 9, 2001 is an influential point and greatly influenced the conclusions. The comparatively lower value of NH_4 at WR3 on Aug. 9, 2001 could be related to some measuring errors. The temporal trend of NH_3 distributions was consistent among the stations.

Table 3.2- 6. Coefficients of determination of NH_3 among stations.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.7304	0.9523	0.8624	0.7847	0.7915
WR3	0.7304	1	0.7508	0.694	0.5544	0.5404
WR4	0.9523	0.7508	1	0.9623	0.7902	0.8076
CRN1	0.8624	0.694	0.9623	1	0.8105	0.8469
BP3	0.7847	0.5544	0.7902	0.8105	1	0.7271
JNB2	0.7915	0.5404	0.8076	0.8469	0.7271	1

Table 3.2- 7. Coefficients of determination of NH_4 among stations after removing the data on Aug. 9, 2001.

R^2	WR2	WR3	WR4	CRN1	BP3	JNB2
WR2	1	0.9887	0.9523	0.8624	0.7847	0.7915
WR3	0.9887	1	0.9771	0.9049	0.8117	0.7885
WR4	0.9523	0.9771	1	0.9623	0.7902	0.8076
CRN1	0.8624	0.9049	0.9623	1	0.8105	0.8469
BP3	0.7847	0.8117	0.7902	0.8105	1	0.7271
JNB2	0.7915	0.7885	0.8076	0.8469	0.7271	1

Assessment of N/P ratio

N/P mass ratios were calculated for each sample to determine the possibility of N-limitation or P-limitation conditions in the aquatic system. Nitrogen mass was calculated as the sum of NH_4, NO_3, and NO_2, whereas the phosphorous was determined as the mass of PO_4. The calculated mean and median values of N/P ratio were shown in Table 3.2-8. Chapra (1997) gave a rough rule of thumb for assessing what nutrient could be the limiting factor based on N/P ration; an N/P ratio value less than 7.2 suggests that nitrogen is limiting and conversely higher values imply that phosphorous will limit algae growth. The calculated mean and median N/P ratio was 0.49 and 0.18 (Table 3.2-8). Hence, the aquatic ecosystem was nitrogen limited in terms of eutrophication.

Table 3.2- 8. Calculated value of N/P mass ratio at the sampling stations.

Site	Sampling size	Mean	Median	Range	Mean N/mean P
WR2	34	0.45	0.13	0.04-5.02	0.27
WR3	29	0.50	0.16	0.03-4.21	0.35
WR4	33	0.50	0.17	0.05-4.35	0.34
WR5	5	1.96	0.54	0.18-4.86	1.32
CRN1	32	0.70	0.21	0.07-8.32	0.50
BP3	28	0.11	0.15	0.05-0.47	0.14
JNB2	28	0.38	0.30	0.11-1.32	0.34
TOTAL	189	0.49	0.18	0.03-8.32	0.33

Cluster Analysis of Water Quality Data

The analysis of water quality data was also extended to the sampling stations, HC1, BC1, BLT1, CC1, JR3, BLC3, and FDB2, in Jourdan River and around bayou area. The same results and conclusion were obtained as the sampling stations in Wolf River and around bayou areas. Hence, the detailed results would not be presented here.

Cluster analysis was first applied to classify the sampling stations of WR2, WR3, WR4, WR5, CRN1, BP3, and JNB2. And then, cluster analysis was applied to the sampling stations of HC1, BC1, BLT1, CC1, JR3, BLC3, and FDB2. The variables used for cluster analysis were the means of the 19 measured physical and chemical parameters, including PH, water temperature, salinity, conductivity, ding whap, stage, flow, DO, TSS, COD, BOD, TOC, TP, PO_4, TKN, NH_4, NO_3, fecal coliform, and chlorophyll a.

The cluster analysis separated the sampling stations in Wolf River and Jourdan River from the stations at the around bayou areas based on the input variables (Fig. 3.3-1 and 3.3-2). The cluster analysis successfully captured the difference between the sampling stations in the river and in the bayou, which could be attributed to the changes in flow regime, human impacts, topography, and soil texture. This could be an indicator that the quality of observed data is of good quality, which clearly reflects the difference of water quality in the river and bayou.

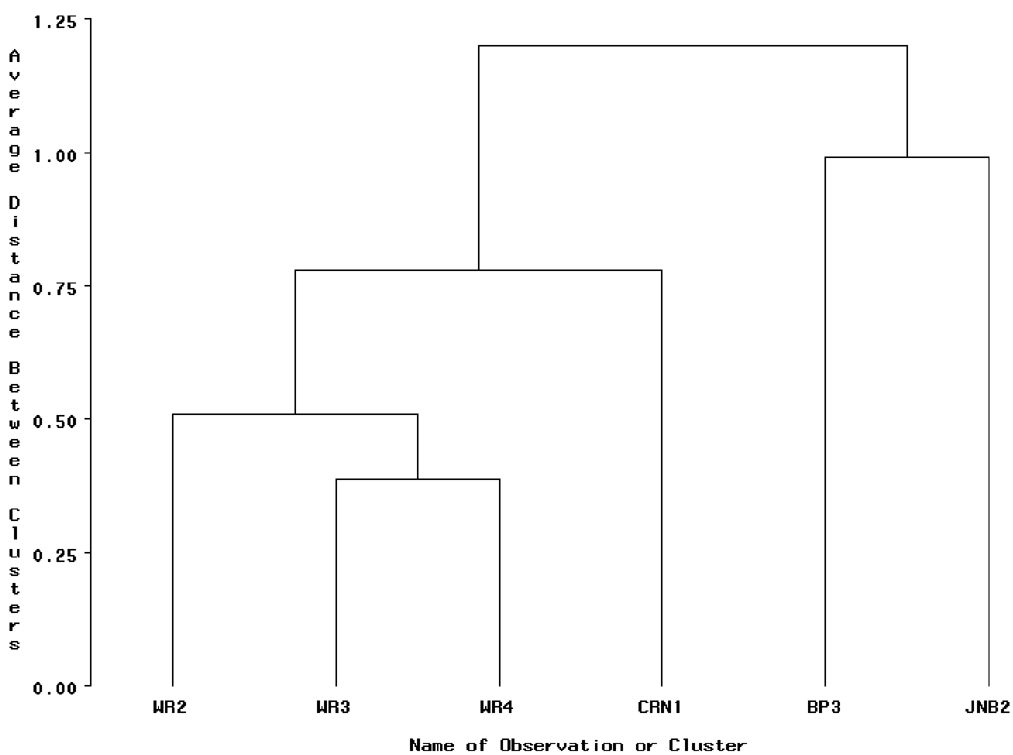


Fig.3.3- 1. Cluster analysis of the sampling stations in Wolf River and around bayou areas.

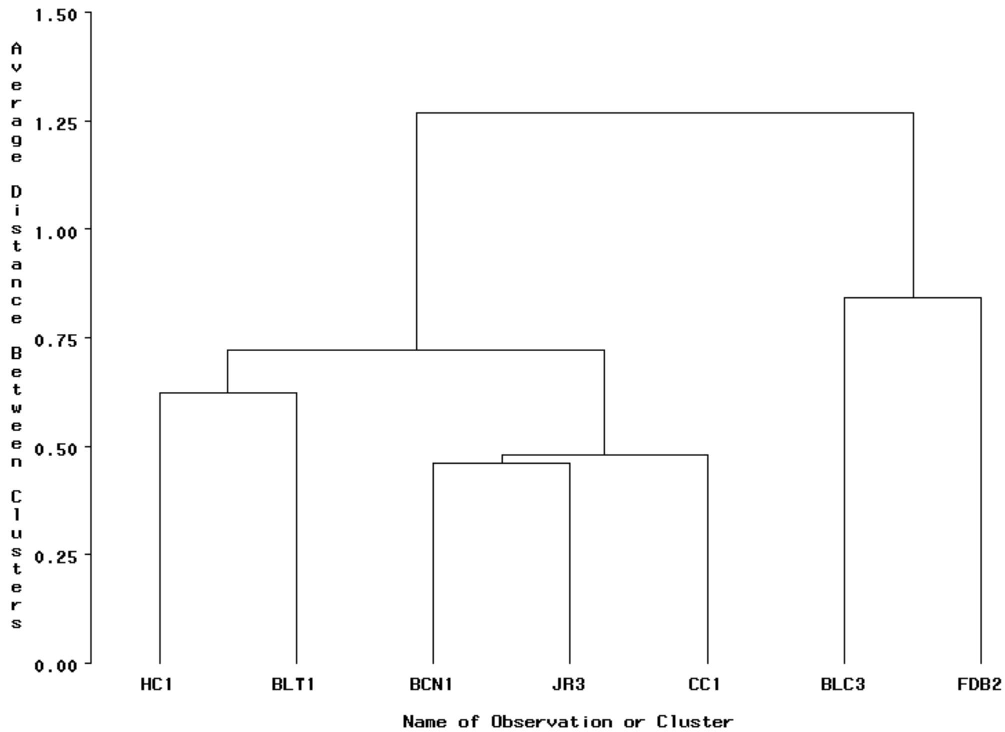


Fig.3.3- 2. Cluster analysis of the sampling stations in Jourdan River and around bayou areas.

Evaluation of the Watershed Delineation

As in prior efforts, the Wolf River watershed was delineated into three sub-watersheds, Reach 1, Reach 2, and Reach 4 (Fig. 3.4-1). The sampling stations, WR2, WR3, WR4, and CRN1, are all located in the sub-watershed Reach 4. The temporal trends of DO, BOD, NO_3, and NH_4, were consistent among these 4 sampling stations, which indicated that the delineation is good enough to capture the spatial variations of these input parameters. However, the temporal trend of PO_4 was not consistent among these 4 sampling stations, and hence, the current delineation is too coarse to capture the spatial variations of PO_4. Since the water quality problem of phosphorus should receive more concerns compared with other constituents, it is recommended that in the future studies, more sub-watersheds should be delineated to reflect the spatial dynamics of phosphorus. However, this will also require higher resolution of field data to support the model development.

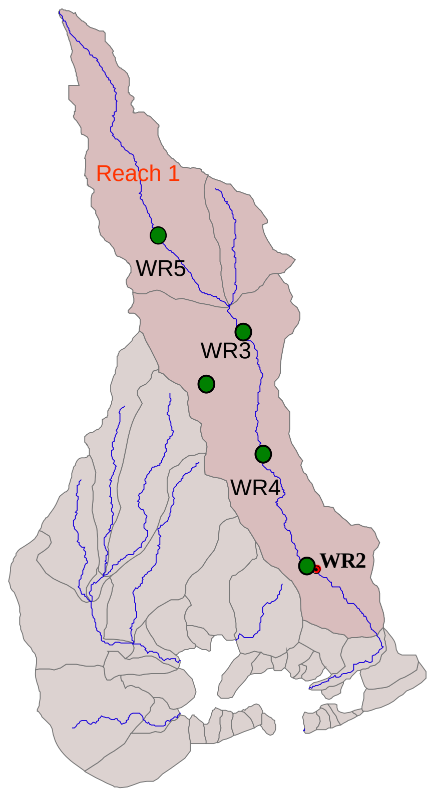


Fig.3.4- 1. Delineation of the St. Louis Bay watershed model.

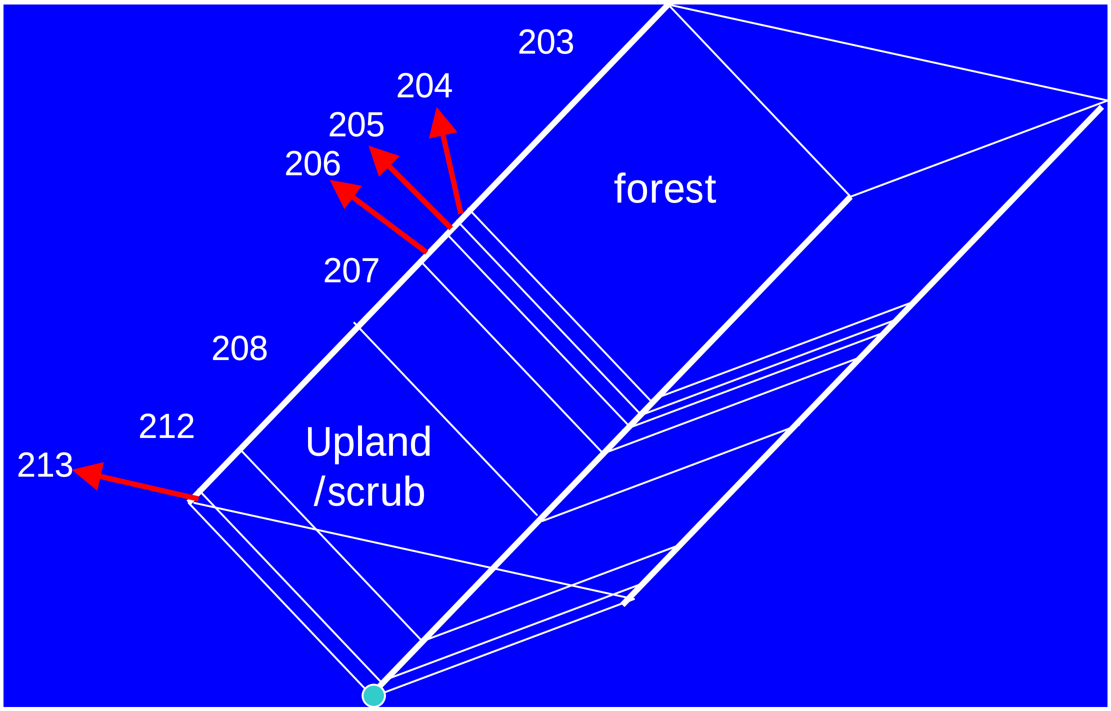
Representation of Spatial Distribution of Landuse by HSPF

The delineated land segments and corresponding area were given in Table 3.5-1. The notation of the land segment has a special meaning for HSPF operations. The first number in the notation indicates the number of delineated sub-watershed. The last two numbers indicate the comparative positions of the land segment; the land segments having lower values are located in the upstream and those having higher values are located in the downstream locations (Fig. 3.5-1). For example, land segment 203 is located upstream to all other land segments, and adjacent to land segment 204.

Table 3.5- 1. The delineated land segments in Reach 2 by BASINS.

Land segment	Land use	Area (acre)
203	Forest	9947
204	Wheat	56
205	Hay	641
206	Soybean	160
207	Pasture	2301
208	Upland/Scrub	5186
212	Wetland	1103
213	Corn	31

Due to the nature of lumped model, HSPF can only output one result from the outlet of each sub-watershed. HSPF is also considered to be a semi-distributed model, since the spatial variations can be captured by delineating more sub-watersheds. Within HSPF, the pollutant loadings from each land segment directly empty into the stream, and there are no interactions on the border line of adjacent land segments, which is not realistic in nature. The land-to-land-to-reach linkage could be established to simulate the nutrient retention effects of forest riparian; however, the modelers have to specify the nutrient concentrations in the down-stream land segments.



Note: This figure was used just for the purpose of illustration and was out of scale.

Fig.3.5- 1. The representation of spatial variation of land segments by HSPF

CHAPTER IV

REFINEMENT OF WATERSHED MODEL DEVELOPMENT

The model development was revisited and refined based on the previous modeling efforts, literature review, and consultation with soil scientists and agronomists of Extension Service of MSU. The primary aspects revisited and refined included fertilization-related input parameters, plant uptake-related input parameters, nutrient input methods, non-crop land simulation using PQUAL module, and recalibration of hydrology in the Jourdan River. Some of the refined model inputs have been substantiated by the St. Louis Bay watershed soil sample data and extensive edge-of-field data collected from related studies. In this chapter, the refinement of model development is given first, and then confirmation of model inputs by soil sample and edge-of-field data is presented.

Refinement of Model Development

The aspects of model development refinement included development of fertilization-related input parameters, evaluation of nutrient input methods, and development of plant uptake-related input parameters, non-cropland simulation using PQUAL module, and recalibration of hydrology in Jourdan River.

Development of Fertilization-related Nutrient Input Parameters

Complex nutrient processes in cropland were modeled using AGCHEM modules within WinHSPF, since cropland is considered to contribute significant amounts of nutrients (Correl et al., 1992). Simulated nutrient processes included fertilization, plant uptake, atmospheric deposition, manure application, and nutrient transformations. An initial calculation of model inputs based on regional crop management practices and recommended fertilizer application rates by Mississippi State University Extension Service (MSU-ES) were selected as model input (Kieffer, 2002). These nutrient inputs reflect the agronomic practices of most recent decade, which is a reasonable approach for watershed simulation studies. However, this may not be the best procedure for long term simulations. Consideration of historical information may provide better estimates of model inputs. Therefore, the objective was to compare the developed nutrient loading functions based on current crop management practices (scenario 1) and those developed through an analysis of information covering the simulation period (scenario 2). Detailed descriptions of nutrient input parameter development for each cropland for scenario 1 were given by Huddleston et al. (2003). The development of nutrient input parameters for scenario 2 based on historical practices are given here.

Development of Nutrient Input Parameters Based on Historical Fertilization Practices

For the Wolf River watershed, cropland was split into four main categories: corn, hay, soybean, and wheat. Hay comprised approximately 78% percent of the total cropland area (Fig. 4.1-1). The nutrient balance within the soil for each crop type vary due to factors such as fertilizer application rates, variations in planting and harvesting dates, and plant uptake of nutrients. The cropland categories were modeled separately so that typical nutrient management practices for each crop could be prescribed.

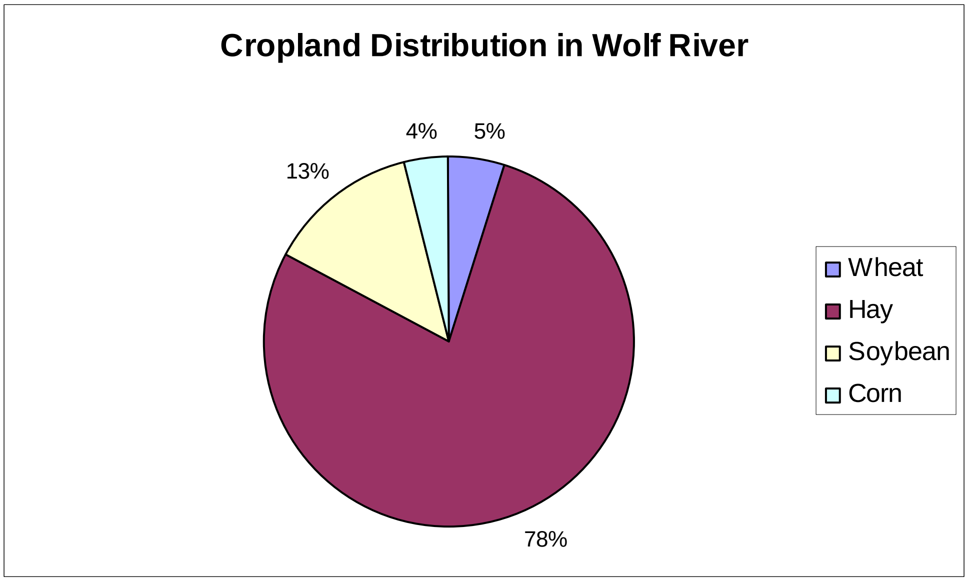


Fig.4.1- 1. Cropland distribution in the Wolf River watershed.

The general steps for determining the nutrient input parameters associated with fertilization practices included calculating the annual nutrient application rates for each crop category, distributing these on a per month basis, and estimating the distribution of monthly nutrients between the surface zone (S.Z.) and the upper zone (U.Z.). The surface zone is a shallow layer of topsoil that is a continuous mixing zone, important for estimating the surface runoff and sediment erosion from the land. The upper soil zone typically corresponds to the depth of incorporation by tillage of applied fertilizer (Donigian, 1976). For the Wolf River watershed, the depths of surface zone and upper zone were assumed to be 0.5 and 6.5 inches from the soil surface, respectively (Huddleston et al., 2003). The basic fertilization techniques include surface broadcasting, soil incorporation, and injection. For modeling purpose, the broadcast nutrient was assumed to be applied into the surface zone, and the injected was assumed to be applied into the upper zone. For the incorporated nutrient, 10% was assumed to be applied into the surface zone, and the remaining 90% was incorporated into the upper zone based on the assumption that incorporation would produce an approximately uniform distribution of nutrient in the top two soil layers (Donigian, 1994).

All nutrient inputs to the model must be in one of the nutrient forms simulated, which include NO_3, NH_4, organic nitrogen, PO_4, and organic phosphorus. For nitrogen application, it was assumed that 25% of the applied nitrogen was in the form of NO_3, and the remaining 75% was in the form of NH_4. The phosphorus fertilizer was assumed to be in the form of PO_4. These assumptions were based on the typical types of fertilizer used in the study area (Kieffer, 2002).

Corn Cropland

The corn yield data used for the modeling period (1965-2001) were obtained from the Mississippi Agricultural Statistics Service (MASS) (Fig. 4.1-2). Yield increases over time, with an average yield of 56.2 bushels/acre. The increased yields are due to development of high yielding varieties, pest control, and more efficient nutrient management. The MSU-ES recommended rate of nitrogen for corn is 1.3 pounds of actual nitrogen for each bushel of yield goal up to 100 bushels per acre (Larson, 2004). Hence, the estimated average annual nitrogen application rate corresponding to a yield goal of 56.2 bushels/acre is calculated to be 73.1 lb/ac. In scenario 1, a nitrogen input value of 100 lb/ac reflected the average corn yield from 1991 to 2001 of 77.1 bushels/acre (Larson, 2004). Typical nitrogen fertilization practices for corn in the study area include one-third of the total applied in March by broadcasting before planting or pre-emergence, and the remainder injected in April (Larson, 2005).

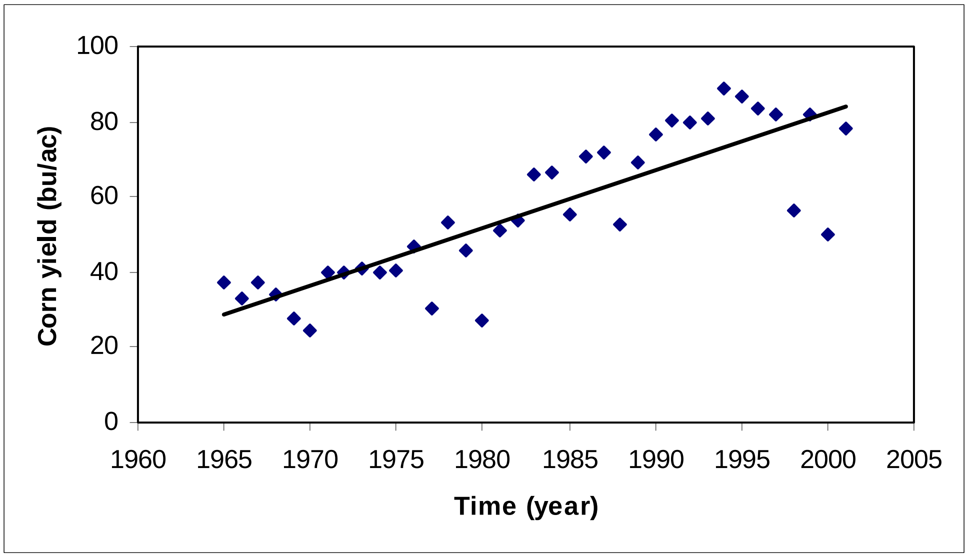


Fig.4.1- 2. Annual corn yield in the study area.

The estimated phosphorus application rates for corn were approximately 17.6 lb/ac, equal to 40 lb P_2O_5 (Larson, 2005). The phosphorus fertilizer was incorporated in November. The estimated annual application rates, temporal and spatial distribution for nitrogen and phosphorus in scenario 1 and 2 are shown in Table 4.1-1.

Table 4.1- 1. Nutrient input parameters of corn cropland for scenarios 1 and 2 (lb/month).

Month	Scenario 1				Scenario 2			
	Nitrogen		Phosphorus		Nitrogen		Phosphorus	
	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.
JAN	-	-	-	-	-	-	-	-
FEB	-	-	-	-	-	-	-	-
MAR	50	-	-	-	24.4	-	-	-
APR	50	-	-	-	-	48.7	-	-
MAY	-	-	-	-	-	-	-	-
JUN	-	-	-	-	-	-	-	-
JUL	-	-	-	-	-	-	-	-
AUG	-	-	-	-	-	-	-	-
SEP	-	-	-	-	-	-	-	-
OCT	-	-	-	-	-	-	-	-
NOV	-	-	1	9	-	-	1.8	15.8
DEC	-	-	-	-	-	-	-	-
Total	100		10		73.1		17.6	

Hay Cropland

In scenario 1, characterization of hay production within the watershed was based on discussions with local county extension agents (Kieffer, 2002). It was assumed that the summer perennial grasses grown were typically 50% bahiagrass and 50% bermudagrass. Ryegrass was the typical winter forage crop grown for cattle grazing. Simulations of bahiagrass, bermudagrass, and ryegrass were all included in hay cropland section. However, it was very important to distinguish between hay and forage cropland. For example, when harvesting pastures for hay most of the nutrients are removed in the crop biomass. With forage nutrients are recycled by grazing animals. Therefore, in scenario 2, ryegrass simulation was not included in the hay cropland section.

In scenario 2, the annual nitrogen application rates were estimated based on an empirical relationship between the amount of applied nitrogen and amount of nitrogen removed by plant uptake. Generally, for hay cropland, the average ratio of the amount of nitrogen removed by plant uptake to that applied by fertilization is about 70% (Watson, 2005). The amounts of nutrients removed by plant uptake were calculated by multiplying the hay crop yield by the percent nitrogen composition in the harvested plant tissues.

State-level data, obtained from Mississippi Agricultural Statistics Service, were used to represent the bermudagrass yield in the Wolf River watershed since county and district-level data were not available. Bermudagrass yields have increased slowly over time, with an average yield of 2.0 tons/acre (Fig. 4.1-3). Based on data from the Arkansas forage database, for bermudagrass, 40 lb of nitrogen is removed by harvesting one ton of forage dry matter (Table 4.1-2). Hence, the amount of nitrogen removed by plant uptake was approximately 80 lb/acre, and the estimated average annual nitrogen application rate during the simulation period was 114.3 lb/acre (Table 4.1-3).

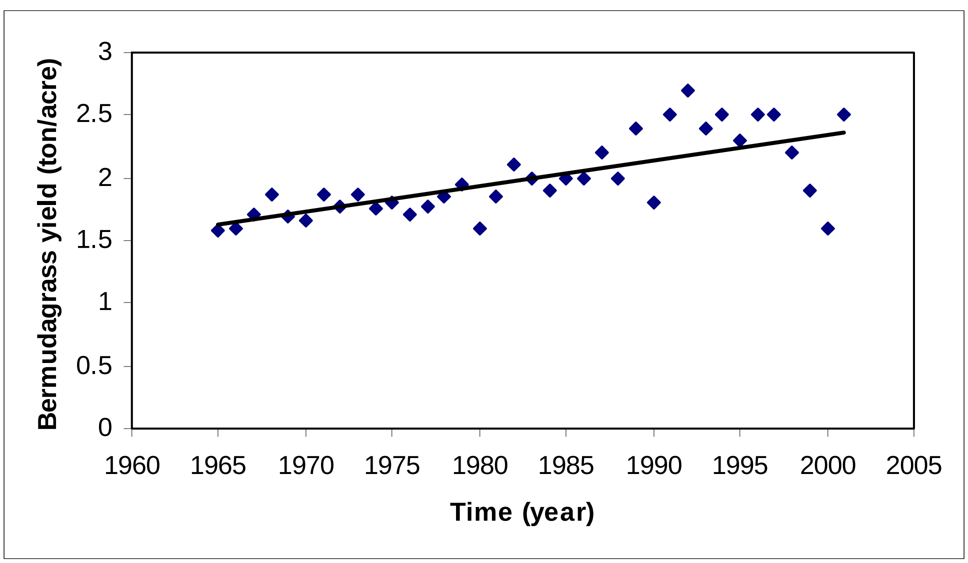


Fig.4.1- 3. State-wide annual bermudagrass yield from 1965-2001 in Mississippi.

Similarly, the estimated average annual nitrogen application rate for bahiagrass was 88.6 lb/acre (Table 4.1-3). The ultimate average annual nitrogen application rate for hay cropland was taken to be the mean of that for bermudagrass and bahiagrass.

Table 4.1- 2. Amount of nutrients removed by per ton of crop dry matter (lb/ac).

Forage	N	P_2O_5	K_2O
Bermudagrass	40	12	44
Bahiagrass	31	8	34
Fescue	36	14	50
Ryegrass	39	16	54
Alfalfa	58	14	56
Legume/grass	39	12	43

* Data were obtained from the Arkansas forage database (1985-1996).

Table 4.1- 3. Estimated nitrogen application rates for bermudagrass and bahiagrass.

Hay	Yield (tons/acre)	Uptake (lb/ac)	Application rate (lb/ac)
Bermudagrass	2.0	80	114.3
Bahiagrass	2.0	62	88.6
Average			101.4

The assumed local fertilization practices for hay was to apply triple thirteen (13-13-13) equally in April, May, and June by broadcasting to meet the crop nitrogen demand of grass (Watson, 2005). Since triple thirteen contains the same amount of nitrogen and phosphorus, the ultimate average annual phosphorus application rate was the same as that of nitrogen. The estimated annual application rates, temporal and spatial distribution for nitrogen and phosphorus in scenario 1 and 2 are shown in Table 4.1-4.

Table 4.1- 4. Nutrient input parameters for hay cropland for scenarios 1 and 2.

Month	Scenario 1				Scenario 2			
	Nitrogen		Phosphorus		Nitrogen		Phosphorus	
	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.
JAN	-	-	-	-	-	-	-	-
FEB	60	-	-	-	-	-	-	-
MAR	-	-	-	-	-	-	-	-
APR	30	30	30	30	33.8	-	33.8	-
MAY	-	-	-	-	33.8	-	33.8	-
JUN	-	-	-	-	33.8	-	33.8	-
JUL	60	-	-	-	-	-	-	-
AUG	-	-	-	-	-	-	-	-
SEP	60	-	-	-	-	-	-	-
OCT	-	-	-	-	-	-	-	-
NOV	30	30	30	30	-	-	-	-
DEC	60	-	-	-	-	-	-	-
Total	360		120		101.4		101.4	

Soybean Cropland

Nitrogen fertilizer is usually not required for soybean because it is a leguminous crop. Generally, the phosphorus application rate for soybean is estimated based on soil phosphorus test levels. Soil test phosphorus categories for Mississippi and recommended fertilizer rates are shown in Table 4.1-5. Soil phosphorus test data were obtained from the Mississippi State University Soil Testing Laboratory (MSU-STL). During the simulation period, only 12 years of data were available for the study area (Table 4.1-6). The calculation of annual phosphorus application rates took into account the percentage of soil samples in each soil test category. For each of the 12 years, the estimated annual application rate was the sum of the product of percentage of soil test category and corresponding recommended application rate (Table 4.1-6). The average application rate of 15.6 lb/acre was used to represent the average annual phosphorus application rate for the whole simulation period. Phosphorus fertilization practices involved a March application that was incorporated into the soil by plowing. The estimated annual application rates, temporal and spatial distribution for nitrogen and phosphorus in scenario 1 and 2 are shown in Table 4.1-7.

It has been reported that soil phosphorus test levels greater than 36 lb/acre resulted in little response to phosphorus fertilization in Mississippi (Hoover, 1968). The phosphorus application rate in modeling scenario 1 of 70 lb/ac was higher than the recommended phosphorus rate for very low soil phosphorus test category of 52 lb/ac. This indicates that only 52 lb/ac phosphorus is needed under the assumption that all the soybean cropland has very low soil phosphorus test levels. However, the soil phosphorus test data indicated that less than 20% of soil samples had very low soil test level of phosphorus (Table 4.1-6). Hence, the annual phosphorus application rate used in the scenario 1 may be an over-estimation.

Table 4.1- 5. Mississippi soil phosphorus test categories and recommended fertilizer rates for soybean (After Varco, 1998).

Soil test category	Soil test phosphorus	Recommended
Very low	0-18	52
Low	19-36	26
Medium	37-72	13
High	73-144	0
Very high	>144	0

Table 4.1- 6. Mississippi soil phosphorus test data for the Wolf River watershed and recommended application rates for soybean (MSU Soil Test Laboratory, 2005).

Year	Sample size	Soil test category (%)				Application rate
		Very low	Low	Medium	High	
1971	415	-	38	28	34	13.5
1972	211	-	22	40	38	10.9
1974	1001	-	32	29	39	12.1
1975	892	13	15	35	37	15.2
1976	1314	10	14	30	46	12.7
1986	1009	5	22	42	31	13.8
1990	737	19	24	35	22	20.7
1991	447	4	16	49	31	12.6
1992	214	20	18	32	30	19.2
1993	300	9	24	41	27	16.3
1994	290	10	18	37	35	14.7
2001	273	38	10	27	24	25.9
Average						15.6

Table 4.1- 7. Nutrient input parameters for soybean cropland for scenarios 1 and 2.

Month	Scenario 1				Scenario 2			
	Nitrogen		Phosphorus		Nitrogen		Phosphorus	
	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.
JAN	-	-	-	-	-	-	-	-
FEB	-	-	-	-	-	-	-	-
MAR	-	-	-	-	-	-	1.6	14.0
APR	-	-	-	-	-	-	-	-
MAY	-	-	-	-	-	-	-	-
JUN	-	-	-	-	-	-	-	-
JUL	-	-	-	-	-	-	-	-
AUG	-	-	-	-	-	-	-	-
SEP	-	-	-	-	-	-	-	-
OCT	-	-	-	-	-	-	-	-
NOV	-	-	7	63	-	-	-	-
DEC	-	-	-	-	-	-	-	-
Total			70				15.6	

Wheat Cropland

Since there were not enough historical yield data to estimate the average nutrient application rates, the recommended fertilization application rates by MSU-ES were used in scenario 2 (Larson, 2005). The recommended nitrogen application rates were 100 lb/ac. A quarter of the nitrogen was assumed to be applied in November by incorporation, and the remainder was applied in February and March by broadcasting (Larson, 2005). A phosphorus rate of 11 lb/acre of phosphorus (25 lb/ac P_2O_5) incorporated in October was the MSU-ES recommended (Larson, 2005). The estimated annual application rates, temporal and spatial distribution for nitrogen and phosphorus in scenario 1 and 2 are shown in Table 4.1-8.

Table 4.1- 8. Nutrient input parameters for wheat cropland for scenarios 1 and 2.

Month	Scenario 1				Scenario 2			
	Nitrogen		Phosphorus		Nitrogen		Phosphorus	
	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.	S.Z.	U.Z.
JAN	-	-	-	-	-	-	-	-
FEB	37.5	-	-	-	37.5	-	-	-
MAR	37.5	-	-	-	37.5	-	-	-
APR	-	-	-	-	-	-	-	-
MAY	-	-	-	-	-	-	-	-
JUN	-	-	-	-	-	-	-	-
JUL	-	-	-	-	-	-	-	-
AUG	-	-	-	-	-	-	-	-
SEP	-	-	-	-	-	-	-	-
OCT	-	-	-	-	-	-	1.1	9.9
NOV	-	-	-	-	-	-	-	-
DEC	2.5	22.5	-	-	2.5	22.5	-	-
Total	100				100		11.0	

Total Annual Nutrient Loading in Scenario 1 and 2

The total annual nitrogen loading from cropland in scenario 1 was 1,261,320 lb/ac compared to 380,950 lb/ac in scenario 2 (Fig. 4.1-4). The decrease in total nitrogen loading was mainly due to removal of the simulation of ryegrass from hay cropland and using application rates for all crops that reflected the entire simulation period (1965-2001). The total annual phosphorus loadings from cropland in scenario 1 (456,620) was slightly higher than that in scenario 2 (359,974.8 lb/acre) (Fig. 4.1-5). The decrease in total phosphorus loading was mainly caused by the differences in scenarios 1 and 2 for hay and soybean fertilization.

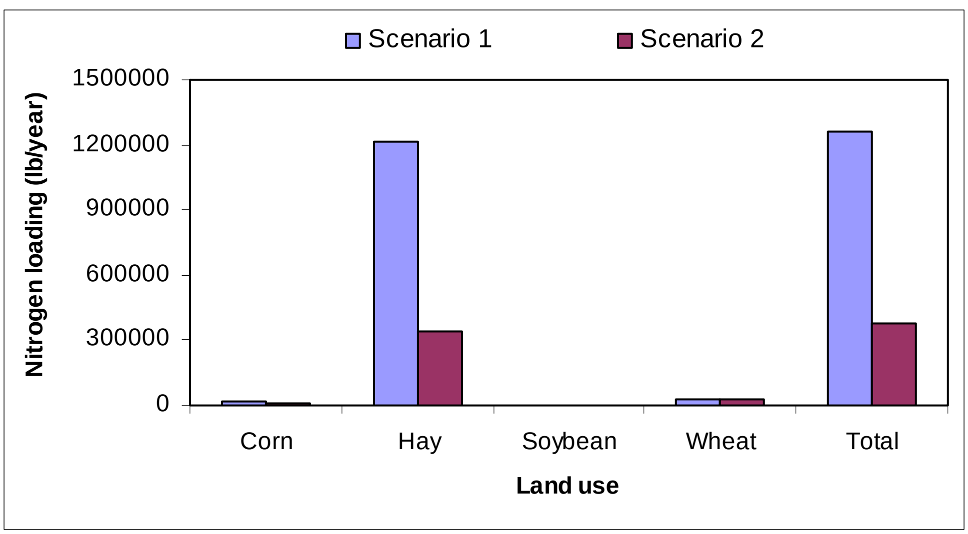


Fig.4.1- 4. Annual cropland nitrogen loadings for scenarios1 and 2.

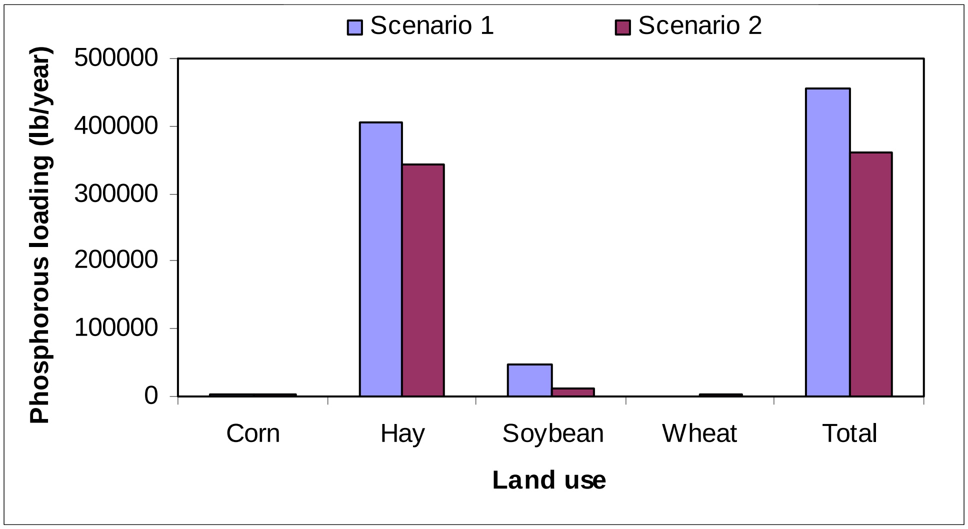


Fig.4.1- 5. Annual cropland phosphorus loadings for modeling 1 and 2.

Evaluation of Two Nutrients Input Methods: Monthly Data Block and Manual Time Series

The preparation of nutrient input dataset into HSPF model is a very-time consuming processes. The nutrients must enter into the model by a specific form; nitrogen must be in the form of NO_3, NH_4, or organic nitrogen, and phosphorus must enter in the form of PO_4 or organic phosphorus. In addition, the nutrients must be specified to input into a particular soil layers, surface layer or upper layer. Further, the fertilization rate of nutrients varies for different crops. Finally, for each sub-watershed, the unit load of nutrients from manure application could be different.

Basically, there are three methods available: Special Action Block, Monthly Data Block and Manual Time Series. In the Special Action Block, the variable could be changed at a specified time to simulate human activities, such as plowing, application of fertilizer and pesticide (Bicknell et al., 2001). However, the specification of input parameters in Special Action Block is very complicated. Hence, this method would not be discussed herein. For the method of Monthly Data Block; the user specifies the monthly rate of input nutrients, and HSPF will generate daily time series by the internal interpolation function. Modelers can also construct the daily time series manually, and then write the time series into the model. Hence, we refer to this method as Manual Time Series.

AGCHEM modules have been successfully applied to simulate the complex watershed water quality processes in several studies (Bicknell et al., 1984; Moore et al., 1988; Donigian et al., 1994; Im, et al., 2003; Filoso, et al., 2004; Saleh and Du, 2004; Liu et al., 2005). Only Moore et al. (1988) mentioned that the nitrogen sources from fertilizer were put into the model using Special Action Block. It is unknown whether the Monthly Data Block will generate the same boundary loadings as the manually constructed daily time series. Hence, the objective is to evaluate the interpolation function of the Monthly Data Block and compare these two input methods based on the developed St. Louis Bay watershed model.

St Louis Bay Watershed Model Nutrient Inputs

For the St. Louis Bay watershed model, the simulated nutrients non-point sources include atmospheric deposition, fertilization practice, and manure application. For modeling purposes, the soils were classified into four layers in the St. Louis Bay watershed and different layers are associated with different flow fluxes (Table 4.1-9). The nutrients from fertilization practices are applied to the surface or upper layer depending on the fertilization methods. The detailed development of fertilization rate of nutrients for wheat, corn, soybean, and hay, was described by Liu et al. (2005). The monitored data from the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) were used to determine the contribution of nutrients from the atmospheric deposition. Since no NADP/NTN stations are located in the study area, the data from the nearest station, LA30, located in Washington Parish, Louisiana were applied to the study area. The nutrients from atmospheric deposition were assumed to be only applied to the surface layer. The unit loads of nutrients from atmospheric deposition and fertilization practice were assumed to be same for all the sub-watersheds. However, the unit load of nutrients from manure application was different among the sub-watersheds depending on the number of cattle (Kieffer, 2002). For simplification, it was assumed that all the produced manure was applied to the hay cropland since the magnitude of manure production is very small compared with fertilization. The phosphorus from manure was assumed to be equally distributed between the surface and upper layer. The phosphorus from manure was assumed to be 50% in the form of PO_4 and 50% in the form of organic phosphorus. The calculated loading functions of PO_4 and organic phosphorus from all these non-point sources for each cropland were shown in Table 4.1-10 and 4.1-11, respectively. The simulation period was from 1965 to 2001. The HSPF model domain considered herein is the Wolf River watershed including the sub-watersheds labeled as 018, 019, and 020.

Table 4.1- 9. Soil layer division for watershed modeling.

Soil Layers	Depth Range (inch)	Associated hydrological and water quality processes
Surface Layer	0 - 0.5	Surface runoff, fertilization, and plant uptake
Upper Layer	0.5 – 6.5	Interflow, fertilization, and plant uptake
Lower Layer	6.5 – 47.5	Evapotranspiration and plant uptake
Groundwater Layer	47.5 – 133.5	Ground water

Table 4.1- 10. Intended input of PO_4 onto the cropland.

Sub-basin Crop	Soil Layer	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Wheat	Surface	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0
Wheat	Upper	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.9	0.0	0.0
Soybean	Surface	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Soybean	Upper	0.0	0.0	14.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corn	Surface	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0
Corn	Upper	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.8	0.0
018 Hay	Surface	0.04	0.04	0.04	33.84	33.84	33.84	0.04	0.04	0.04	0.04	0.04	0.04
018 Hay	Upper	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
019 Hay	Surface	0.03	0.03	0.03	33.83	33.83	33.83	0.03	0.03	0.03	0.03	0.03	0.03
019 Hay	Upper	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
020 Hay	Surface	0.06	0.06	0.06	33.86	33.86	33.86	0.06	0.06	0.06	0.06	0.06	0.06
020 Hay	Upper	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06

Table 4.1- 11. Intended input of organic phosphorus onto the cropland.

Sub-basin Crop	Soil Layer	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
018 Hay	Surface	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
018 Hay	Upper	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
019 Hay	Surface	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
019 Hay	Upper	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
020 Hay	Surface	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
020 Hay	Upper	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06

Evaluation of Interpolation Function of the Monthly Data Block

It is very easy to use the Monthly Data Block to enter the nutrients into the HSPF model. A monthly-table needs to be constructed first to specify the daily application rate of nutrients for each month, and then link this table to a specific pervious land. HSPF uses a linear function to interpolate the daily nutrient input based on the given values for the start of this month and next month. The interpolation function is given by Equation (4.1-1).

DAYVAL= MVAL1 + (MVAL2 - MVAL1)*(RDAY - 1)/RNDAYS

Equation (4.1-1)

where, DAYVAL: the interpolated amount of nutrients for a particular day

MVAL1: applied amount of nutrients at the start of this month

MVAL2: applied amount of nutrients at the start of next month

RDAY: day of the month

RNDAY: number of days in this month

A simple modeling scenario was devised to evaluate this interpolation function. For the hay cropland in the sub-watershed 018, it is assumed that the PO_4 is applied only in March, with daily application rate of 3.0 lb/day (Table 4.1-12). Hence, the annual intended total input of PO_4 is 93.0 by multiplying the daily rates and the total days in March.

Table 4.1- 12. The devised test of PO_4 application.

Month	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Daily Rate			3.0									

The generated PO_4 boundary condition of water year 1965 for the selected hay cropland was shown in Fig. 4.1-6. Obviously, Monthly Data Block distributes the PO_4 to the previous month by the interpolation function even though the users do not intend to. Hence, the Monthly Data Block can misrepresent the intended temporal distribution of applied nutrients by agricultural management practices. In addition, the calculated sum of the generated daily PO_4 loading was 88.54 lb, not equal to the intended application rate of 93 lb. Monthly Data Block can not preserve the users' intended mass of input nutrients. The difference between the generated boundary loadings and the intended input loadings depends on the difference in the numbers of days between this month and previous month.

In order to examine the effects of application timing on generated boundary loadings, 12 modeling scenarios were devised; the daily application rate of 3 lb was applied to January, February, March, April, May, June, July, August, September, October, November, and December, respectively. The application timing has impacts on the generated loadings by the interpolated function with the errors ranging from underestimation of 1.582% to overestimation of 5.408% (Table 4.1-13). Once there are more days in this month than the previous month, Monthly Data Block under-estimates the boundary loadings, and in the reverse situation, Monthly Data Block over-estimates the boundary loadings (Table 4.1-13). The magnitude of the errors depends on the magnitude of difference in the number of days between this month and previous month (Table 4.1-13).

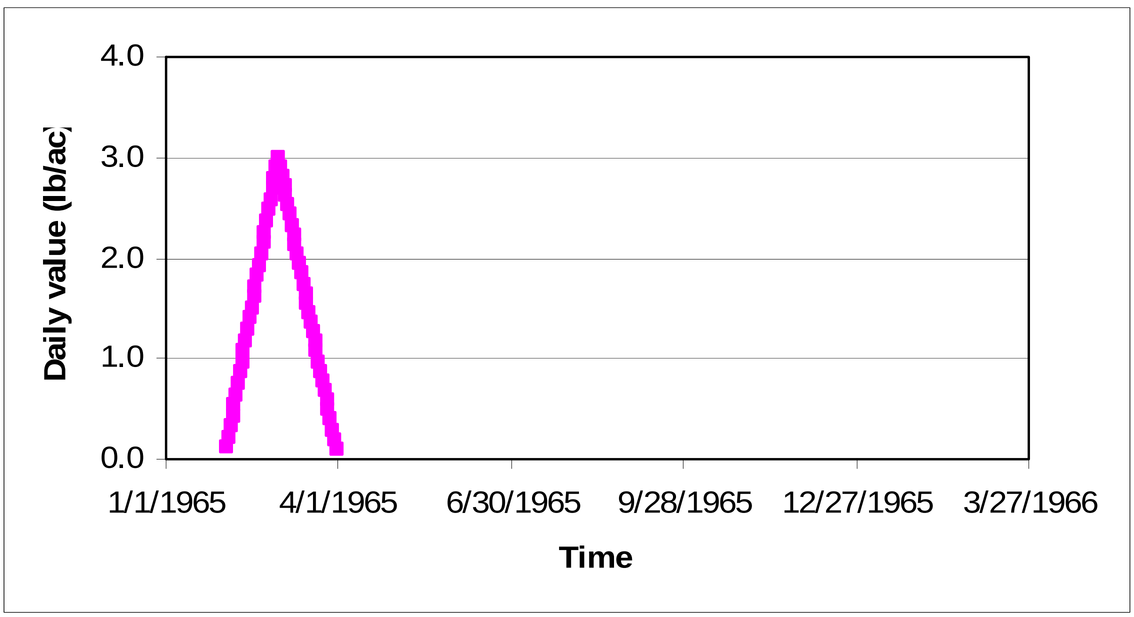


Fig.4.1- 6. Generated PO_4 boundary loading conditions by using Monthly Data Block.

Table 4.1- 13. The errors in boundary loadings introduced by Monthly-Data block.

Month	Intended (lb/ac)	Generated (lb/ac)	Error (lb/ac)	Percentage (%)
JAN	93	93.0336	+0.0336	+0.036
FEB	84	88.5432	+4.5432	+5.408
MAR	93	88.5432	-4.4568	-4.792
APR	90	91.5288	+1.5288	+1.699
MAY	93	91.5288	-1.4712	-1.582
JUN	90	91.5288	+1.5288	+1.699
JUL	93	91.5288	-1.4712	-1.582
AUG	93	93.0336	-0.0336	+0.036
SEP	90	91.5288	+1.5288	+1.699
OCT	93	91.5288	-1.4712	-1.582
NOV	90	91.5288	+1.5288	+1.699
DEC	93	91.5288	-1.4712	-1.582

For the St. Louis Bay watershed model, 12 monthly-data tables have been constructed to enter the PO_4 loadings from the croplands to the model and 6 monthly-data tables have been established to input the organic loadings (Table 4.1-10 and Table 4.1-11). Considering the cropland area and simulation duration, the intended PO_4 loadings will be over-estimated by 65,134.75 lbs by the interpolation function of the Monthly-Date block (Table 4.1-14). For a large agriculture-intensive watershed, the generated errors in boundary loadings by Monthly Data Block could be high enough to affect the reliability of the developed watershed model.

Table 4.1- 14. The errors in generated boundary condition for St. Louis Bay watershed model.

Landuse	Area (acre)	Error in generated nutrients input by Monthly Data Block (lb)
Wheat	253	-1866.38
Soybean	693	-18680.00
Corn	87	+981.74
Hay-018*	2169	+38290.31
Hay-019*	641	+15205.44
Hay-020*	1214	+31203.64
Total	5057	+65134.75

*018, 019, and 020 indicate the three delineated sub-watersheds in Wolf River.

Another disadvantage of the Monthly Data Block option is that the interpolation function will automatically distribute the monthly application rate to daily rate. However, the fertilizer is often applied once a month, or twice a month, or at most weekly.

Comparison of Modeling Performance by Using Monthly Data Block and Manual Time Series

Two modeling scenarios were devised by constructing two Manual Time Series to compare with the Monthly Data Block. For modeling scenario 1, the developed monthly phosphorus boundary loadings were equally distributed from monthly rate to daily rate. For modeling scenario 2, the developed phosphorus boundary loadings were assumed to be applied once at the middle of the month (the 15th day of the month) to simulate the actual fertilization practice. The modeling performances were compared for water year 2000 in the Wolf River watershed.

4.1.2.3.1 Construction of Manual Time Series

The time series can be constructed manually by specifying the application rate for each day over the simulation duration. The method of Manual Time Series is very flexible and is able to simulate the daily, weekly, or monthly application practice. However, the preparation of input dataset is very time-consuming, especially for long-time simulation. To simulate daily application practice of nutrients, the monthly application rate has to be converted to daily rate. For different month, the different numbers have to be used to convert from monthly rate to daily rate. In addition, for long-time simulation period, the leap year has also to be considered to avoid input errors. Further, many input time series have to be constructed. For the developed St. Louis Bay watershed model, 18 time series have to be established for PO_4 and organic phosphorus. Finally, the prepared spreadsheet has to be converted to a particular format in order to import into the WDM project file. VBA\Excel is a comparatively simple and useful tool to help prepare the dataset by creating some MACROs to simplify the repeated processes. The steps of constructing the manual daily series were 1) create several MACROs using VBA/Excel to generate 18 input time series for phosphorus; 2) create a script to read the generated time series into WDM file; 3) establish linkage between the constructed time series and the corresponding land segments.

In the St. Louis Bay watershed, hay cropland contributes much more phosphorus loadings compared with wheat, corn, and soybean cropland. The area of hay cropland is much larger than the other three croplands (Table 4.1-14) and the unit fertilization rate of phosphorus is also higher than the other three croplands (Table 4.1-10 and 4.1-11). Hence, generated phosphorus boundary loadings from hay cropland were compared to examine the difference in model inputs between Monthly Data Block and Manual Time Series.

Modeling Scenario 1

In modeling scenario 1, monthly application of PO_4 in April, May, and June, was equally distributed into daily rates by Manual Time Series (Fig. 4.1-7). There are nearly no differences in the generated PO_4 loading boundary conditions in April and May by Monthly Data Block and Manual Time Series. However, for Monthly Data Block method, nearly half of the applied PO_4 intended for June was distributed to March (Fig. 4.1-7). Hence, Monthly Data Block artificially created PO_4 inputs in March and decreased the PO_4 loadings in June by half (Fig. 4.1-7).

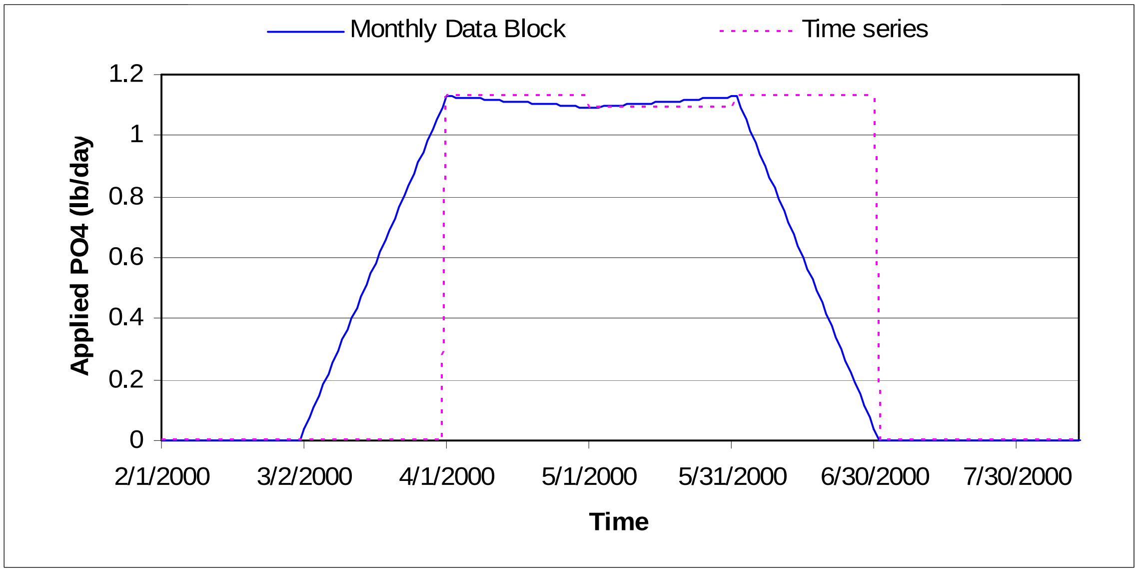


Fig.4.1- 7. Comparison of generated PO_4 input by using Monthly Data Block and Manual Time Series in modeling scenario 1.

The developed watershed model responded very well to the differences in generated PO_4 boundary loading conditions by using Monthly Data Block and Manual Time Series. Before March, there were nearly no differences in simulated PO_4 by using Monthly Data Block and Manual Time Series (Fig. 4.1-8). The simulated in-stream PO_4 concentrations by Monthly Data Block were higher than Manual Time Series in March and lower than Manual Time Series in June (Fig. 4.1-8). This illustrates that the overall model is responsive to the manner in which loads are applied and demonstrates the importance of understanding and developing good model application practices.

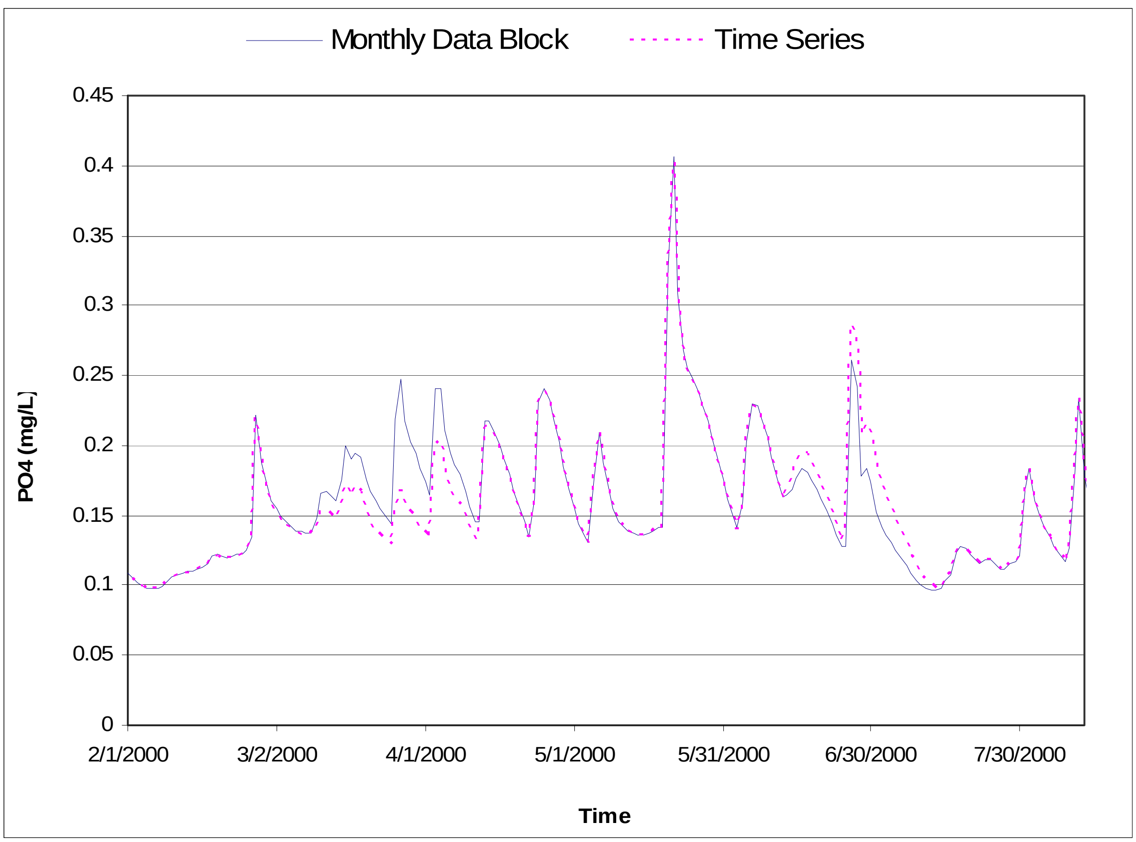


Fig.4.1- 8. Comparison of simulated PO_4 by using Monthly Data Block and Manual Time Series in modeling scenario 1.

Modeling scenario 2

In modeling scenario 2, the time series was manually constructed to put the monthly application of PO_4 on one day, the middle of the month, to reflect the field monthly fertilizer application practice (Fig. 4.1-9). For the Manual Time Series method, there is no PO_4 loading until April 15, but three high single-daily PO_4 inputs on April 15, May 15, and June 15 (Fig. 4.1-9). The highly different generated PO_4 boundary loadings by these two methods resulted in the much differences in modeled in-stream PO_4 simulations (Fig. 4.1-10). The simulated PO_4 concentrations by Manual Time Series were systematically lower than by Monthly Data Block from March 1 to April 15 (Fig. 4.1-10). The field fertilization practices were simulated very well by Manual Time Series; three high peak PO_4 simulations responded to three high single daily application of PO_4 (Fig. 4.1-10). In order to simulate peak events, care must be taken to provide adequate model detail such as using an appropriate load application scenario.

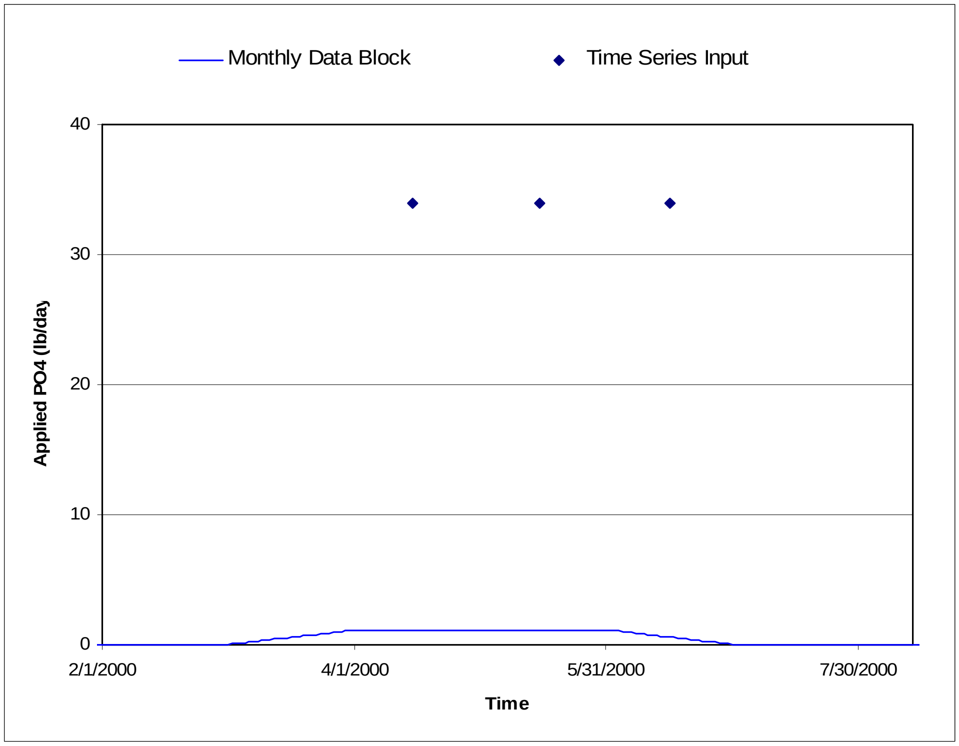


Fig.4.1- 9. Comparison of generated PO_4 input by using Monthly Data Block and Manual Time Series in modeling scenario 2.

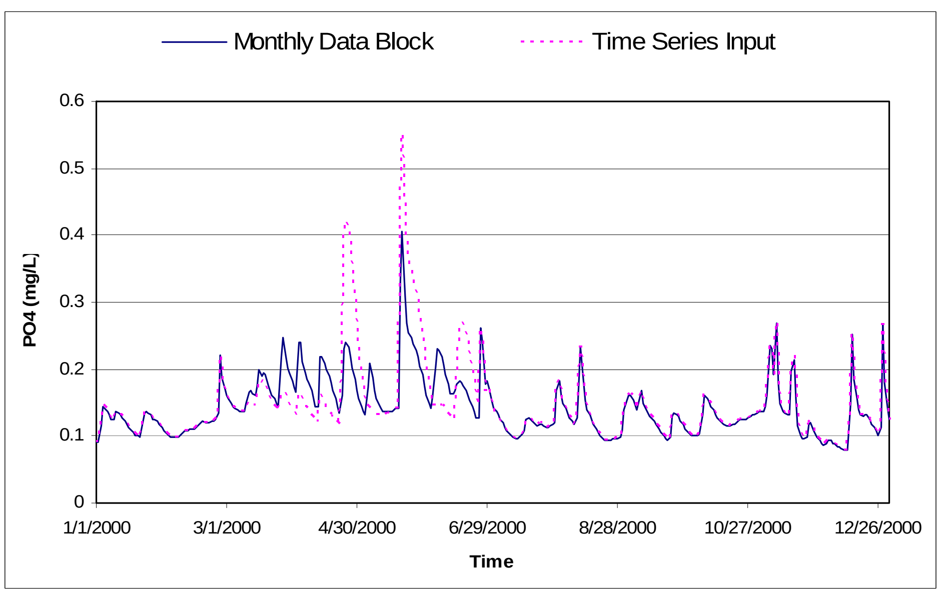


Fig.4.1- 10. Comparison of simulated PO_4 by using Monthly Data Block and Manual Time Series in modeling scenario 2.

Simulation of Plant Uptake Process Using AGCHEM

Plant uptake is the processes that plants absorb nutrients from the soil to satisfy theirs needs for growth. The amount of nutrients that plant can uptake varies at different stages of plant growth. The majority of uptake occurs in the root zone area. The plant uptake together with nutrient applications are the dominant portions of the nutrient balance (Donigian et al., 1994).

There are three options available to simulate the process of plant uptake: Mischaelis-Menten, first order kinetic rate, and yield-based algorithm (Bicknell, 2001). The Mischaelis-Menten method is primarily devised to simulate forest land. Since only croplands were simulated using AGCHEM modules in the St. Louis Bay watershed, this method will not be discussed here. The yield-based algorithm is a better approach in representing the nutrient management practices than the first order kinetic rate method in that it is developed to allows the crop needs to be satisfied and less sensitive to soil nutrient level (Bicknell et al., 2001). However, the application of yield-based algorithm requires much more field data than first order kinetic rate method. The necessary information required for applying yield-based algorithm includes crop yield, plant growth in the temporal and spatial distribution, and percentage of nutrients in the crop dry weight. The extensive data requirement limits the application of yield-based algorithm in simulating plant uptake. The first-order kinetic method has been widely used to simulate plant uptake (Moore et al, 1988; Donigian et al., 1994; Im et al., 2003; Filoso et al., 2004; Saleh and Du, 2004).

The objective of this section is to compare the algorithms of first order kinetic and yield-based methods, develop the plant uptake input parameters using yield-based algorithm, evaluate the key parameter of Plant Uptake Target, and assess the impacts of uptake ratio of NO_3 to NH_4 on plant uptake, and NO_3 and NH_4 outflow fluxes.

Comparison of Algorithms between First Order Kinetic Rate and Yield-Based Method

The modeling of nitrogen plant uptake using first order kinetics is described by Equation (4.1-2) and (4.1-3). Plants can utilize both NO_3 and NH_4; hence, the nitrogen uptake has to be distributed between these two forms. NO3UTF and NH4UTF indicate the fractions of nitrogen uptake for NO_3 and NH_4, respectively, and the sum of NO3UTF and NH4UTF must equal to unity. Obviously, the calculation of plant uptake is a direct function of soil nutrient level, which makes the simulation of plant uptake process too sensitive to nutrient application rate. Though plant growth is affected by soil nutrient level, the nutrient uptake is not a direct function of available nutrients. The amount of nutrient removed by plant uptake at the early growing season is very small, but the soil nutrient concentrations are often very high due to fertilization. Table 4.1-15 concluded the first order rates of plant uptake from several studies.

$$UTNI = [NO3] * k * NO3UTF \qquad \text{Equation (4.1-2)}$$

where, UTNI = plant uptake of NO_3 (lb/ac)

[NO3] = storage of NO_3 in the soil (lb/ac)

k = first order kinetic rate of plant uptake (per day)

NO3UTF = fraction of nitrogen uptake in the form of NO_3

$$UTAM = [AMSU] * k * NH4UTF \qquad \text{Equation (4.1-3)}$$

where, UTAM = plant uptake of NH_4 (lb/ac)

[AMSU] = storage of NH_4 in the soil (lb/ac)

k = first order kinetic rate of plant uptake (per day)

NH4UTF = fraction of nitrogen uptake in the form of NH_4

Table 4.1- 15. Summarized first order rates of plant uptake.

Studies	Filoso, et al., 2004 (/day)			Moore, et al. (1988) (/day)	Im et al., (2003) (/day)
	Surface layer	Upper layer	Lower layer		
JAN	0.00	0.00	0.00	Surface layer: 0.0-0.46 Upper layer: 0.0-0.46 Lower layer: 0.0-0.2	Surface layer: 0.35-0.55 Upper layer: 0.35-0.60 Lower layer: 0.1-0.2 Groundwater layer: 0.05
FEB	0.00	0.00	0.00		
MAR	0.50	0.50	0.00		
APR	0.50	0.55	0.05		
MAY	0.50	0.45	0.10		
JUN	0.40	0.45	0.15		
JUL	0.50	0.55	0.15		
AUG	0.50	0.40	0.10		
SEP	0.25	0.25	0.10		
OCT	0.15	0.10	0.00		
NOV	0.15	0.10	0.00		
DEC	0.00	0.00	0.00		

The first order kinetic rate of plant uptake is dependent on soil temperature. The first order rate at 35 ˚C is considered to be optimal rate, and first order rate at soil temperature above 35 ˚C is assumed to be at optimal rate (Bicknell, 2001). For the first order rates below 35 ˚C, a temperature correction coefficient is used to determine the first order rate using Equation (4.1-4). The model will stop simulating plant uptake process under the conditions of extremely low soil moisture and soil temperature below 4 ˚C.

$$k = k_{35}\theta^{(T-35.0)} \qquad \text{Equation (4.1-4)}$$

where k = first order kinetic rate of plant uptake at soil temperature T ˚C (per day)

k_{35} = first order kinetic rate of plant uptake at soil temperature 35 ˚C (per day)

Θ = temperature correction coefficient

T = soil layer temperature ˚C

The algorithm used by yield-based method is described by Equation (4.1-5). The yield-based algorithm is able to simulate the temporal distributions of plant's nutrient requirement at different growth stages by using NUPTFM to distribute the annual uptake to monthly uptake. In addition, the spatial distribution of plant uptake could be modeled by specifying NUPTM in the different soil layers. Different from first order kinetic method, the input parameters of the yield-based algorithm have physical meaning and can be developed using observed field data. The determination of annual plant uptake can be derived from crop yield and nutrient composition. The monthly fraction of total annual uptake, NUPTFM, can be determined from crop growth curve. The soil layer fraction of monthly uptake, NUPTM, can be estimated based on crop root distribution data or literature review. However, for the first order kinetic method, the development of input parameters is based on calibration until an expected nutrient balance is reached. Hence, the yield –based algorithm provides a better approach to simulate plant uptake process.

$$MONTGT = NUPTGT * NUPTFM(MON) * NUPTM(MON) * CRPFRC(MON, ICROP)$$

Equation (4.1-5)

where, MONTGT = monthly plant uptake target for current crop (lb/ac)

NUPTGT = total annual uptake target (lb/ac)

NUPTFM = monthly fraction of total annual uptake target

NUPTM = soil layer fraction of monthly uptake target

CRPFRC = fraction of monthly uptake target for current crop

MON = current month

ICROP = index for current crop

Development of Input Parameters Using Yield-based Algorithm

The major input parameters needed to be developed include the total annual uptake target (NUPTGT), monthly fraction of NUPTGT (NUPTFM), and soil layer fraction of MONTGT (NUTPM). The development of these parameters was discussed separately. Development of NUPTGT

The total annual uptake target, NUPTGT, was developed based on the crop yields and nutrient composition in the dry weight. The average annual yield data of corn, wheat, soybean, and hay used for the modeling period (1965-2001) were obtained from the Mississippi Agricultural Statistics Service (MASS) (Table 4.1-16 and 4.1-17). The yield data of hay is reported in dry weight. However, the yield data for corn, wheat, and soybean have to be converted from fresh weight to dry weight by converting factors (Table 4.1-16). The nitrogen and phosphorus compositions in the dry weight for the crops were obtained from the Agricultural Waste Management Field Handbook (Table 4.1-16 and Table 4.1-17) (USDA, 1992). The calculated nitrogen and phosphorus annual uptakes for corn, wheat, and soybean were given in Table 4.1-16. The annual nitrogen uptake of soybean was set as zero since there was no nitrogen input from fertilization and the nitrogen fixation process was not simulated in this study.

Table 4.1- 16. Calculated annual nutrient uptake targets for wheat, corn, and soybean.

Crop	Yield bu/ac	Conversion factor (lb/bu)	Dry weight (lb/ac)	Percent N in dry weight (%)	Percent P in dry weight (%)	Annual N uptake (lb/ac)	Annual P uptake (lb/ac)
Wheat	30.9	60	1,854	2.08	0.26	38.6	4.8
Corn	56.1	56	3,142	1.61	0.28	50.7	8.8
Soybean	23.0	60	1,380	6.25	0.64	0	8.8

The estimated nitrogen and phosphorus annual uptakes for hay cropland were assumed to be the average value of bahiagrass and bermudagrass, and were provided in Table 4.1-17.

Table 4.1- 17. Calculated annual nutrient uptake targets for hay.

Hay	Dry weight (ton/ac)	Percent N in dry weight (%)	Percent P in dry weight (%)	Annual N uptake (lb/ac)	Annual P uptake (lb/ac)
Bahiagrass	2.0	1.27	0.13	50.8	5.2
Berbudagrass	2.0	1.88	0.19	75.2	7.6
Average				63.0	6.4

Development of NUPTFM

For modeling purpose, the estimated total annual uptake, NUPTGT, has to be distributed to monthly rate to capture nutrients removed by plant uptake at different growth stages. The monthly fraction of total annual uptake is determined based on typical planting dates and crop growth stages in the costal region of Mississippi. Kieffer (2002) initially estimated monthly fraction of annual uptake targets for corn, wheat, and soybean, based on the results provided by Agricultural Waste Management Field Handbook (USDA, 1992). The developed NUPTGT for corn, wheat, soybean, and hay were displayed in from Fig. 4.1-11 to Fig. 4.1-14 (Kieffer, 2002).

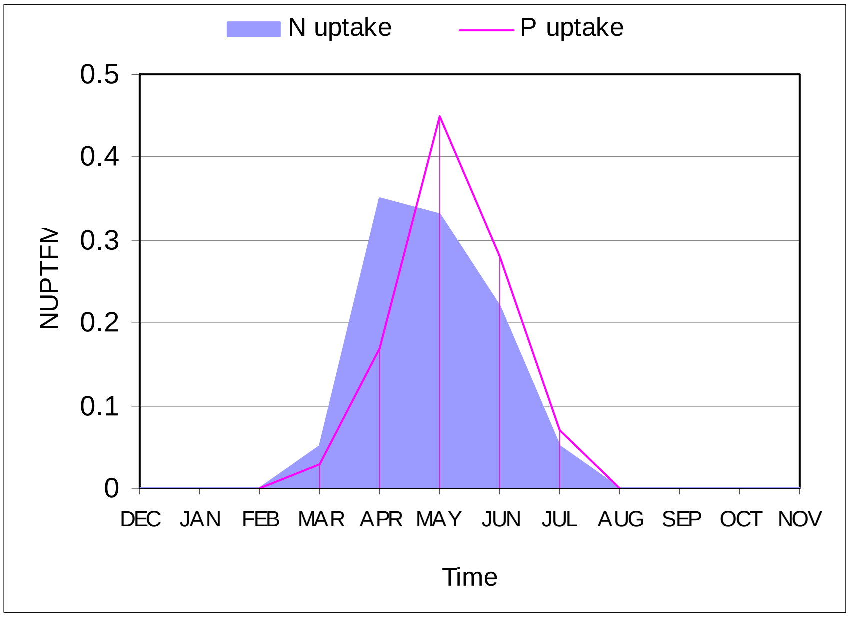


Fig.4.1- 11. The developed NUPTGT for corn cropland.

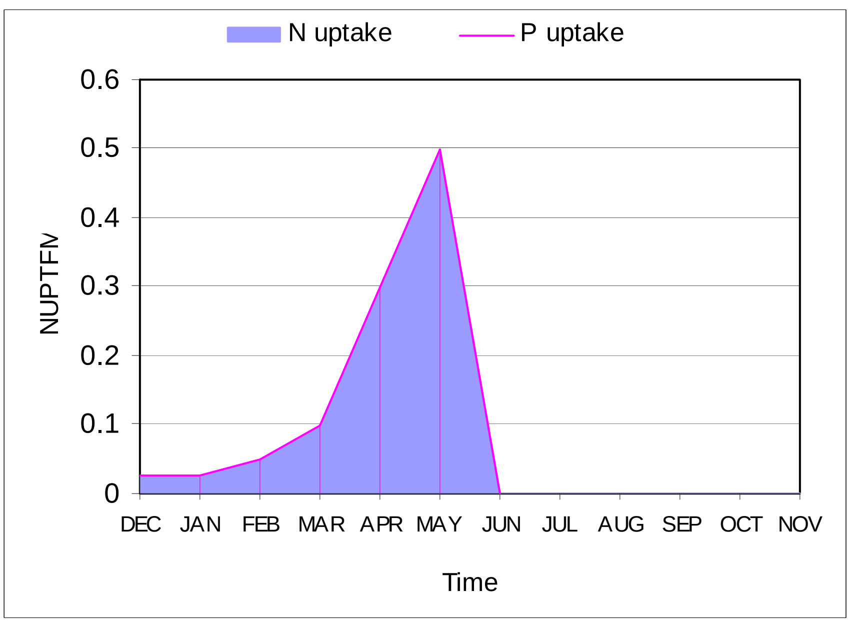


Fig.4.1- 12. The developed NUPTGT for wheat cropland.

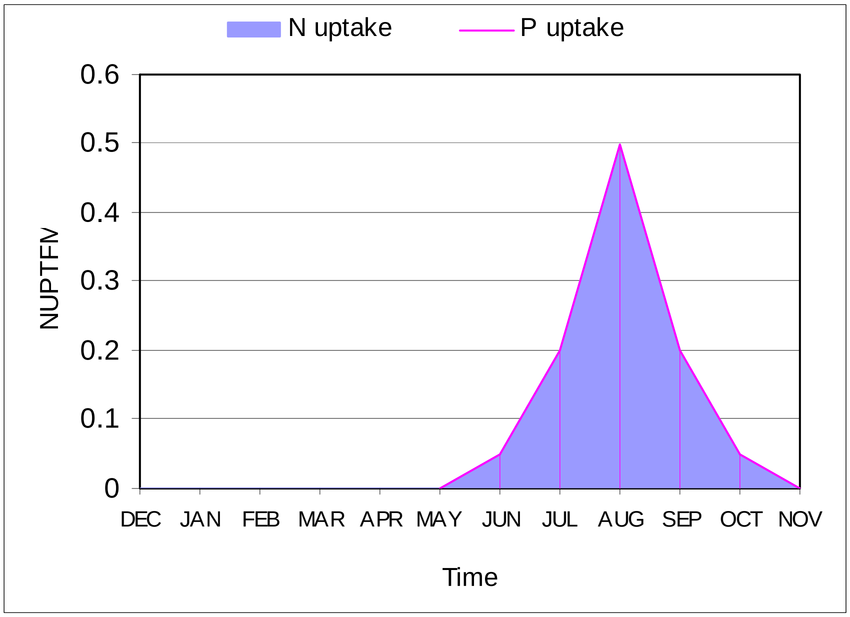


Fig.4.1- 13. The developed NUPTGT for soybean cropland.

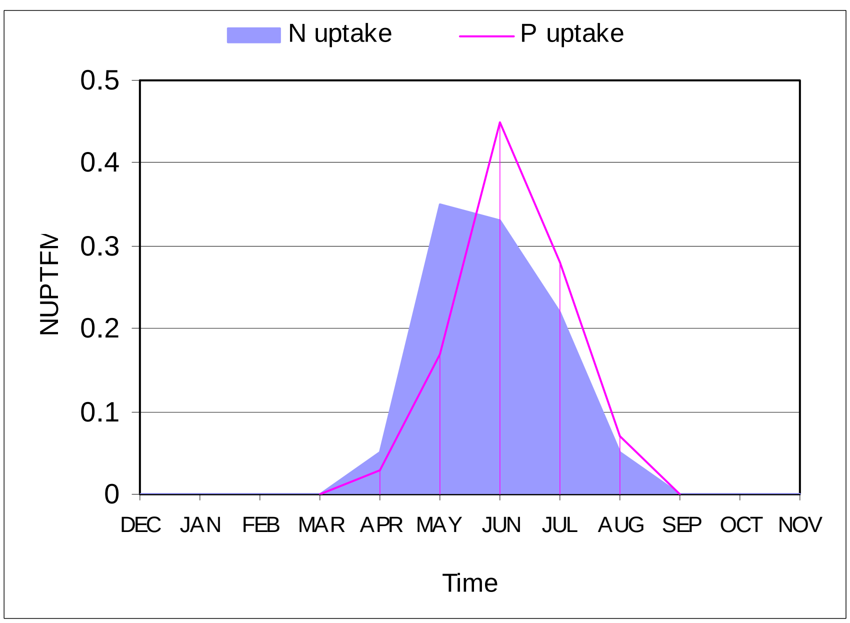


Fig.4.1- 14. The developed NUPTGT for hay cropland.

The typical relationship between growth and nutrient uptake by corn crops developed by Iowa State University Extension Service (ISUES) were shown in Fig.4.1-15 and Fig. 4.1-16 (ISU, 1997). The symbols of Ve through R6 represent the growth stages of the corn (Fig.4.1-15 and Fig. 4.1-16). The initially developed NUPTFM for corn does not match the results provided by ISUES. Since the results from ISUES are more reliable, the NUPTFM was modified to capture the temporal distribution of nutrient uptake by corn (Fig. 4.1-17 and 4.1-18). The amount of simulated plant uptake depends on the soil nutrient level and soil moisture. The shift in the monthly uptake curves between the two scenarios has significant impacts on the simulated nutrient outflow fluxes. Fig.4.1-19 displayed the impacts of shift in the nitrogen monthly uptake curves on nitrogen outflow fluxes from the corn cropland in 2000 in subwatershed 020. For hay cropland, the simulation of winter ryegrass was removed as indicated when developing the nutrient loading scenario 2 for hay cropland. Hence, the monthly fractions for summer month were adjusted to reflect the current simulation (Fig. 4.1-20 and Fig.4.1-21).

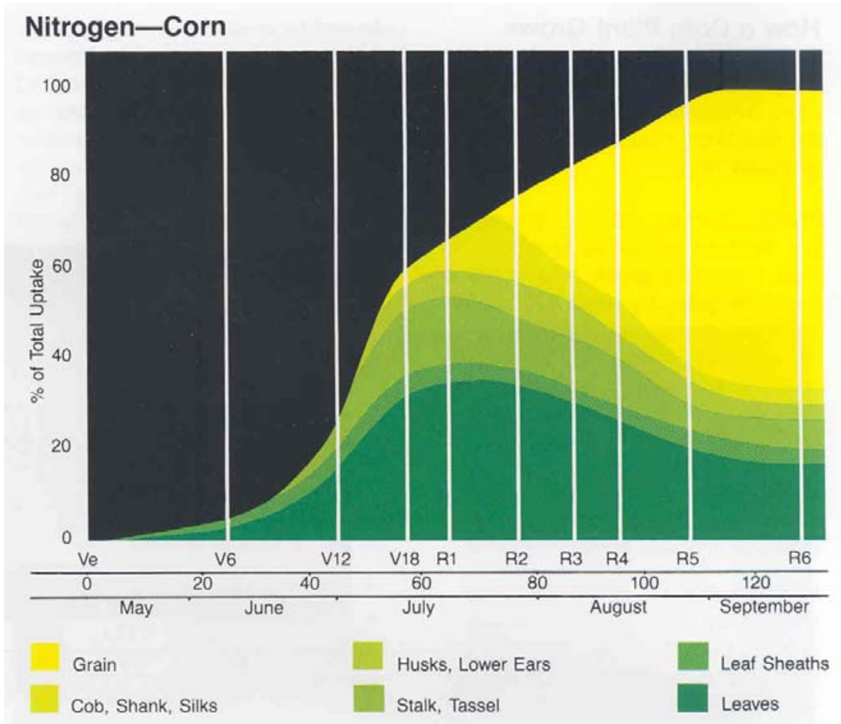


Fig.4.1- 15. Temporal distribution of nitrogen uptake by corn (After ISU, 1997).

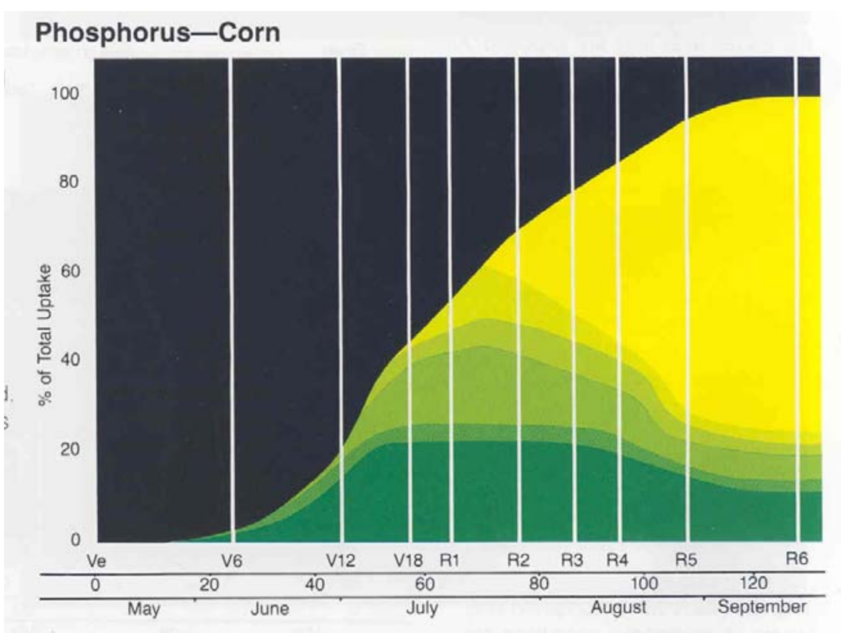


Fig.4.1- 16. Temporal distribution of phosphorus uptake by corn (After ISU, 1997).

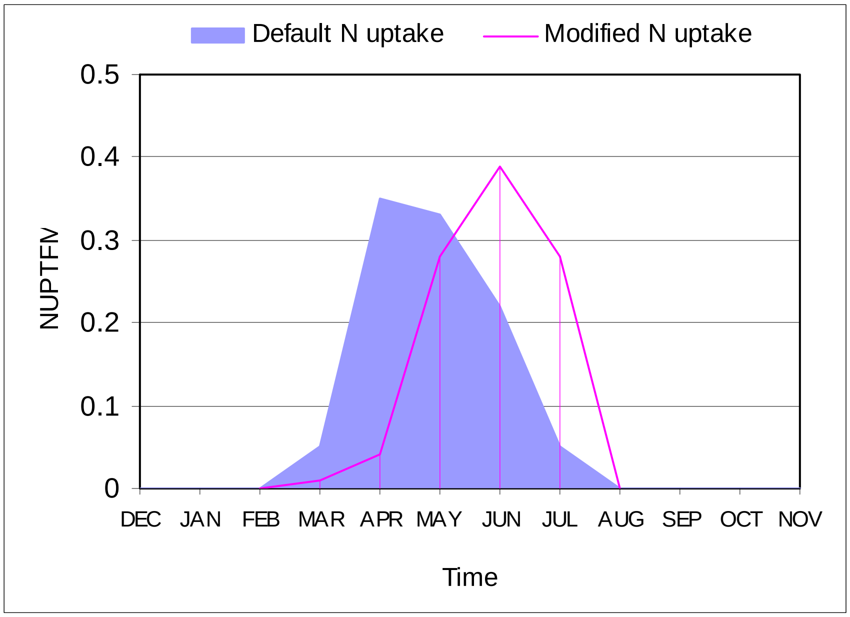


Fig.4.1- 17. Comparison of default and modified nitrogen uptake by corn.

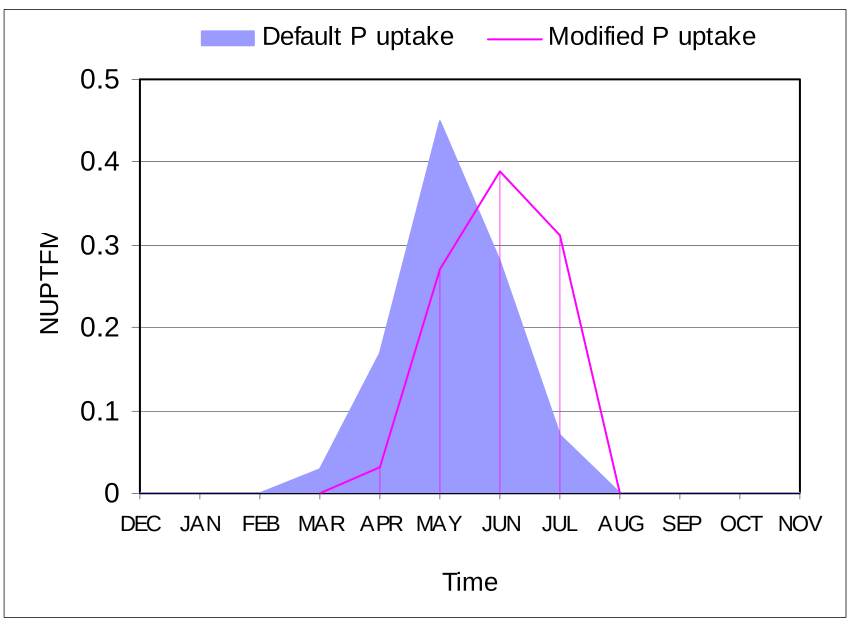


Fig.4.1- 18. Comparison of default and modified phosphorus uptake by corn.

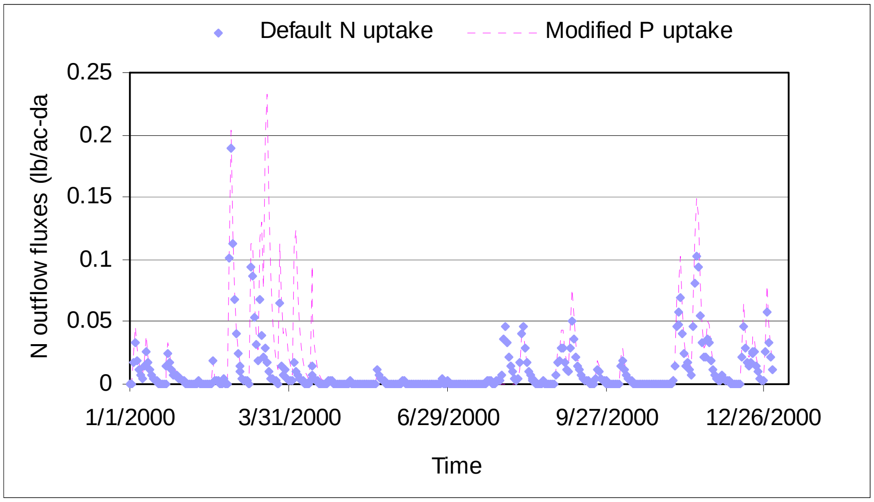


Fig.4.1- 19. Impacts of monthly nutrient uptake rates on simulation of nitrogen outflow fluxes from corn land.

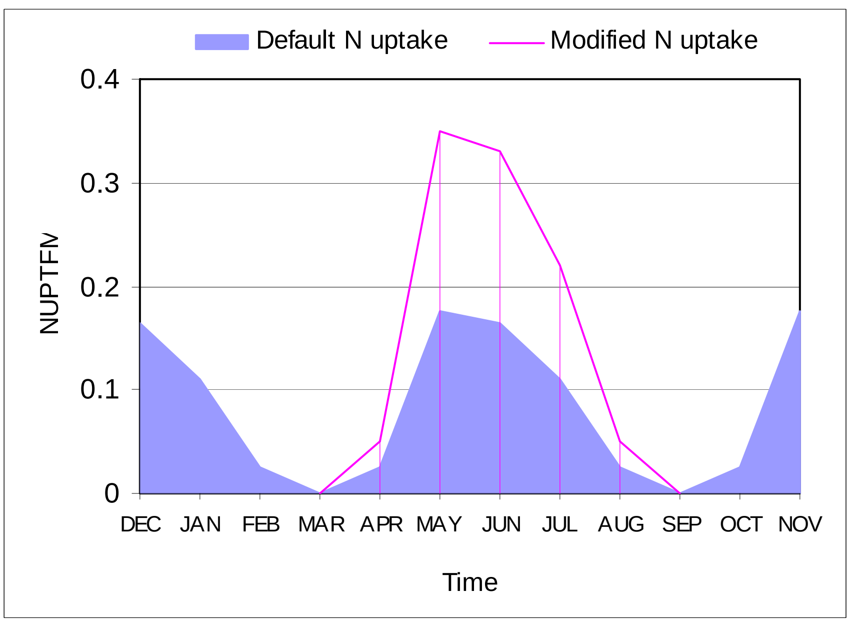


Fig.4.1- 20. Comparison of default and modified nitrogen uptake by hay

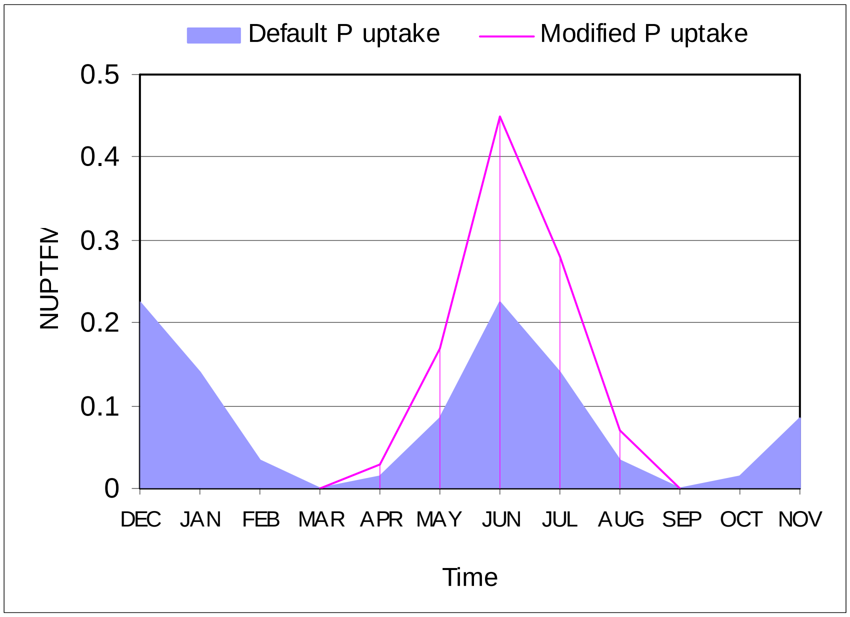


Fig.4.1- 21. Comparison of default and modified phosphorus uptake by hay

Development of NUTPM

For modeling purposes, the monthly nutrient uptake has to be distributed to the four soil layers. The soil layer fraction of monthly uptake rate, NUTPM, was developed based on the crop growth stage and the typical depth of crop root. The assumed depth of groundwater is deeper than the depth of the root zone for all four crops; hence, the amount of uptake in the groundwater zone was set as zero. It was also assumed the spatial distributions of uptake are same for nitrogen and phosphorus. The initial estimation of NUTPM was developed by Kieffer (2002). Minor adjustments, such as decreasing the fraction number in the surface layer, were made to better represent the spatial pattern of plant uptake. The resulting values of NUTPM for corn, wheat, soybean, and hay are summarized in Figs. 4.1-22 to Fig. 4.1-25.

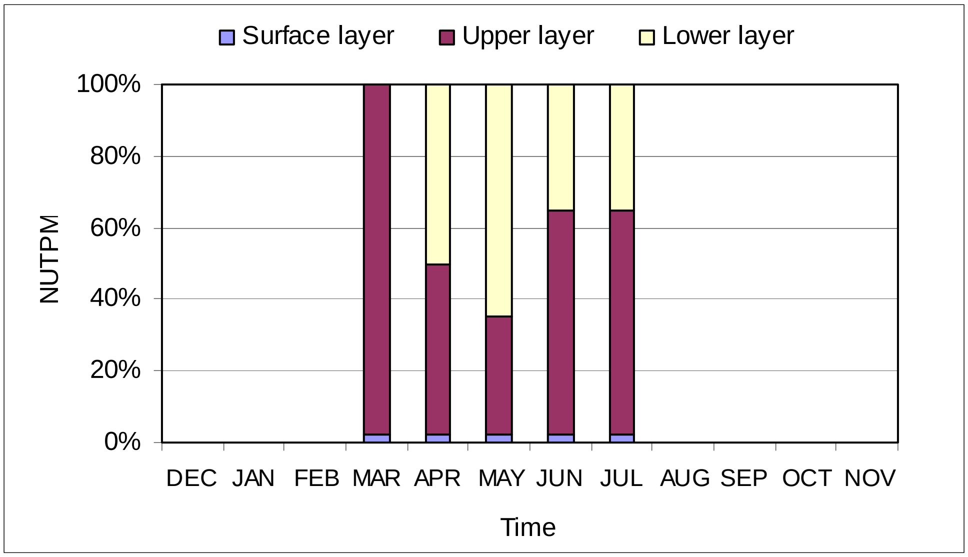


Fig.4.1- 22. The developed NUTPM for corn cropland.

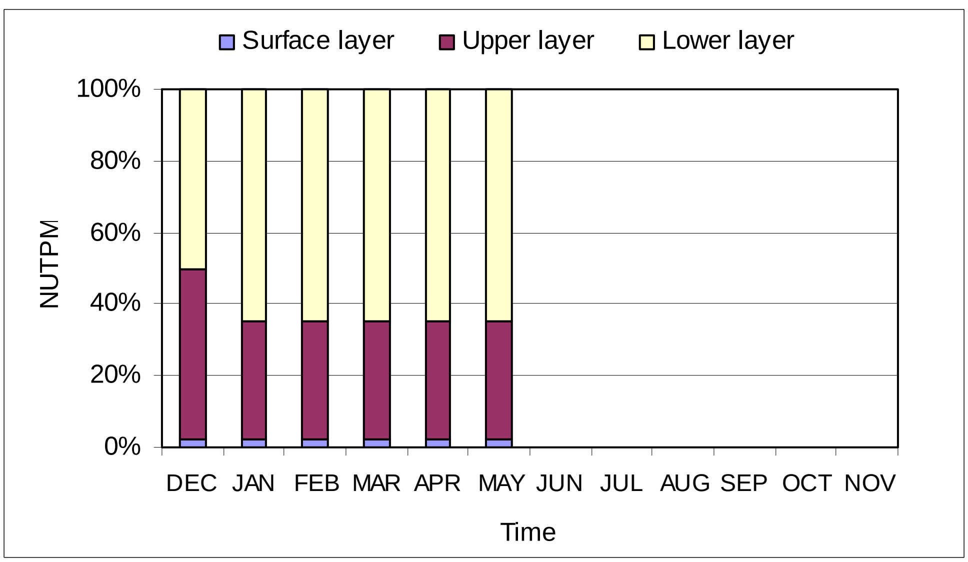


Fig.4.1- 23. The developed NUTPM for wheat cropland.

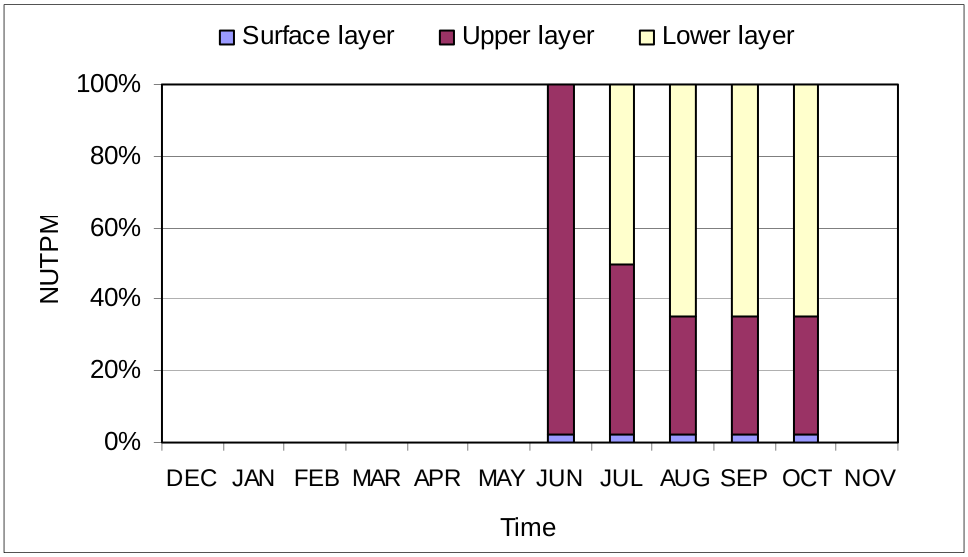


Fig.4.1- 24. The developed NUTPM for soybean cropland.

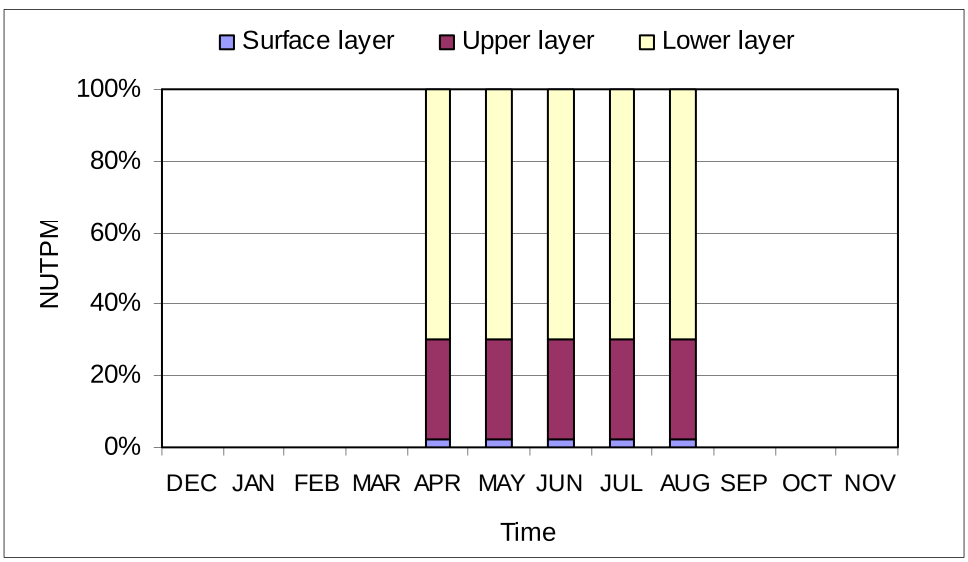


Fig.4.1- 25. The developed NUTPM for hay cropland.

Evaluation of NUPTGT

Since plant uptake is one of the most important nutrient sink processes, it is very important to make sure the correct input of nutrient uptake. Since the parameter NUPTGT was developed based on the crop dry weight and nutrient composition, the estimated plant uptake is the actual amount of nutrient removed from the cropland.

A simple modeling scenario was devised to examine if the generated plant uptake by HSPF is equal to the input parameter of NUPTGT, the intended amount of plant uptake by the modelers. Land segment 104 in the Wolf River watershed was selected to run the model. The modeled constituent is NO_3. For the purpose of simplification, the assumptions were made as follows:

- The annual application of NO_3 is 100 lb/ac.
- The applied NO_3 were equally distributed among April, May, and June.
- Plant uptake ratio of NO_3 to NH_4 is 1 to 0.
- The intended amount of plant uptake, NUPTGT, is 70 lb/ac.
- All the transformation coefficients are set as zero.

The generated value of average annual plant uptake over the entire simulation period (1965-2001) was 56.4, not equal to the input value of NUPTGT (70 lb/ac). To further evaluate the relationship between the intended plant uptake NUPTGT and the generated plant uptake, 8 tests were run with different levels of NUPTGT. The input values of NUPTGT for the 8 test were 7, 17.5, 35, 52.5, 100, 140, 350, and 700 lb/ac.

The generated uptakes under different levels of NUPTGE were given in Fig. 4.1-26. Obviously, the NUPTGT was a target value, not equal to the input values by modelers

(Fig. 4.1-26). When the value of NUPTGT was less than 40 lb/ac, the NUPTGT was overestimated, whereas the NUPTGT was underestimated when values of NUPTGT were higher than 40 lb/ac (Fig. 4.1-26). For some very high value input of NUPTGT, such as 700 lb/ac, the generated plant uptake was approaching the amounts of nutrient available in the soil, 100 lb/ac in this case. The generated amount of plant uptake depends on the soil nutrient level and soil moisture condition (Bicknell et al., 2001). Hence, the parameter of NUPTGT has to be calibrated to the intended amount of plant uptake.

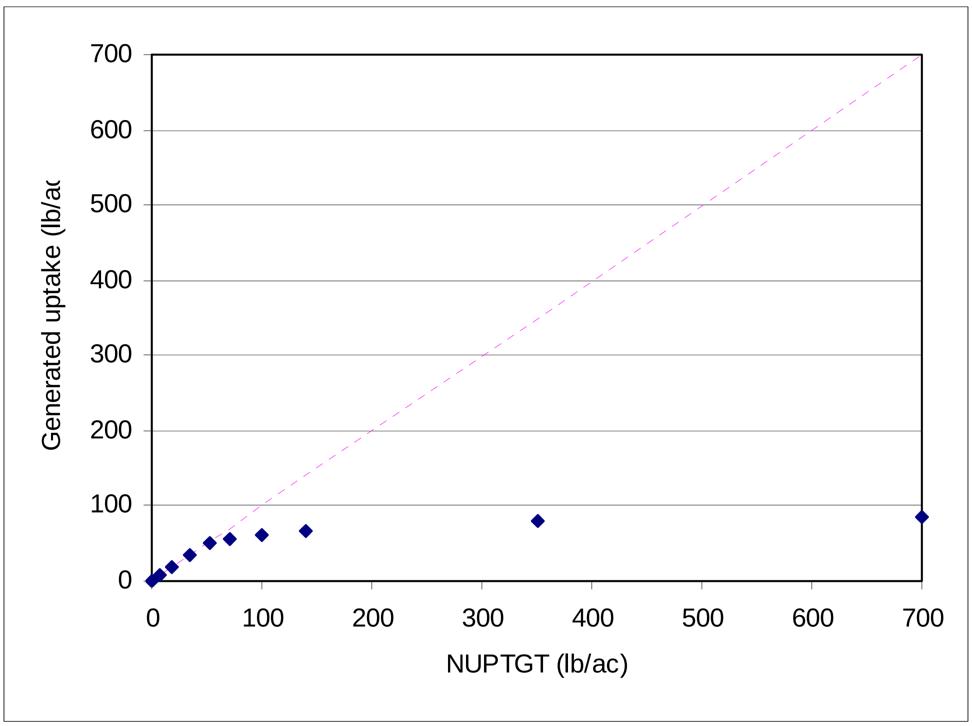


Fig.4.1- 26. Relationship between NUPTGT and generated plant uptake by HSPF.

Impacts of Uptake Ratio of NO_3 to NH_4 on Plant Uptake and Nutrient Outflow Fluxes

The nitrogen forms that plant can uptake include NO_3 and NH_4. Generally, most crops prefer NO_3 instead of NH_4. Five modeling scenarios with different uptake ratios of

NO_3 to NH_4 were designed to evaluate their impacts on the generated plant uptake (TPLNT), NO_3 fluxes (PONO3), and NH_4 fluxes (PONH4) from the land segment based on the developed St. Louis Bay watershed model. The five uptake ratios between NO_3 and NH_4 are 1.0:0.0, 0.75:0.25, 0.5:0.5, 0.25:0.75, and 0.01:0.99. The modeling results from hay cropland 105 were used for analysis.

The nitrogen uptake ratios of NO_3 to NH_4 had strongest impacts on the generated plant uptake ranging from 21.83 to 36.94 lb/ac, but the least effects on NO_3 fluxes from the land segments ranging from 8.86 to 8.98 lb/ac (Fig. 4.1-27). There is minimal difference between uptake ratio of 1.0:0.0 and 0.75 to 0.25 in simulating TPLNT, PONO3 and PONH4 (Fig. 4.1-27). For the St. Louis Bay watershed model, the nitrogen uptake ratio was assumed to be 1.0:0.0. Since some crops could uptake some NH_4 and most crops prefer to NO_3, the nitrogen uptake ratio was adjusted to 0.75:0.25

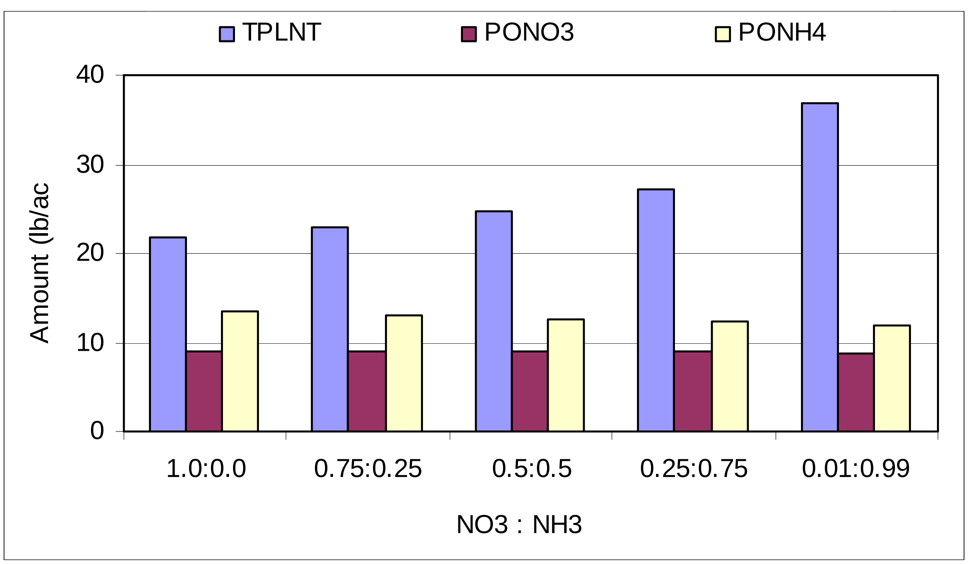


Fig.4.1- 27. Impacts of uptake ratios of NO_3 to NH_4 on TPLNT, PONO3, and PONH4.

Non-cropland Pollutant Simulation Using PQUAL

In addition to AGCHEM modules, HSPF provides an alternative method, PQUAL to simulate pollutant transportation in the pervious land segments. The pollutant transport in the non-crop lands in the St. Louis Bay watershed was simulated using PQUAL. The simulated water quality constituents include BOD, NO_3, PO_4, ORN, and ORP. The water quality constituents are simulated independently based on a simple relationship with water or sediment. Different from AGCHEM module, PQUAL is not able to simulate the complex nutrient processes.

In the PQUAL and IQUAL modules, water quality constituent in the surface outflow was simulated based on basic accumulation and depletion rates together with depletion by washoff. The storage of constituents on the land surface is calculated using Equation (4.1-6) to account for the accumulation and removal processes occurring independent of overland flow, such as atmospheric deposition, cleaning, decay, wind erosion, and deposition (Bicknell et al., 2001).

$$SQO = ACQOP + SQOS * (1.0 - REMQOP) \qquad \text{Equation (4.1-6)}$$

where SQO = storage of available quality constituent on the land surface ($kgha^{-1}$), ACQOP = accumulation rate of the constituent on the land surface ($kgha^{-1}$-day^{-1}), SQOS = SQO at the start of the interval, and REMQOP = unit removal rate of the stored constituent (day^{-1}).

The estimation of input parameter ACQOP is critical for water quality modeling using PQUAL module. The total amount of water quality constituents available for washoff process depends on the accumulation rate, ACQOP. The importance of accurate estimation of ACQOP to water quality modeling using PQUAL is similar to that of the accuracy of precipitation data to hydrology modeling.

The amount of washoffed water quality constituents from the land surface is determined by Equation (4.1-7) (Bicknell et al., 2001). The washoffed water quality constituent in surface runoff is a function of the pollutant storage, the surface outflow of water, and the susceptibility of the quality constituent to washoff.

$$SOQO = SQO * (1.0 - e^{(-SURO*WSFAC})) \qquad \text{Equation (4.1-7)}$$

where SOQO = washoff of the quality constituent from the land surface ($kgha^{-1}hr^{-1}$), SQO = storage of available quality constituent on the land surface (kg/ha), SURO = surface outflow of water ($cmhr^{-1}$), and WSFAC = susceptibility of the quality constituent to washoff (cm^{-1}).

The accumulation rate of water quality constituents on the land surface, ACQOP, for each land use category was calculated based on literature values from a variety of storm water quality studies (Harper, 1994; Maidment, 1993). The results of storm water quality studies are often provided in terms of event mean concentration. For application of such data, it was necessary to convert the constituent concentrations to an accumulation rate. This was accomplished by multiplying the mean concentration by an estimated annual runoff volume (Harper, 1994). The annual runoff volume can be estimated by the product of the annual average rainfall and runoff coefficients. The initial estimations of the accumulation rates of water quality constituents were done by Kieffer (2002) (Table 4.1-18). Due to the undeveloped nature of St. Louis Bay watershed, the accumulation rates of BOD from forestland and upland scrub were assumed negligible.

Table 4.1- 18. Initial Estimations of ACQOP rate for each land use (Kieffer, 2002).

ACQOP (lb/ac-year)	Forest	Upland scrub	Pasture	Wetland	Pervious urban	Impervious urban
BOD	0.0	0.0	26.6	15.3	22.9	103.3
NO3	0.89	0.89	5.18	2.65	1.64	7.39
ORN	0.89	0.89	7.77	2.65	3.79	17.05
PO4	0.049	0.049	1.82	0.43	0.16	0.74
ORP	0.049	0.049	0.66	0.20	0.61	2.74

The above estimated parameters of ACQOP were directly entered into the model to simulate the pollutant movement in the non-croplands (Kieffer, 2002). However, the input pollutant accumulation rate, ACQOP, is a model-specific parameter, and could be much different even between adjacent watersheds (Bicknell, 2005). The values of ACQOP should be calibrated to match the observed data (Bicknell, 2005). There is no recommended value range for the input parameter of ACQOP, and ACQOP is landuse-specific. If we just blindly increase or decrease the value of ACQOP for the six landuses to calibrate the model, the calibration activity would become a number-game. A calibration methodology was developed by calculating the relative ratios of ACQOP among the landuses, and then simultaneously increasing and decreasing the values of ACQOP for all landuses by keeping the relative ratio constant during the calibration processes.

Recalibration of Hydrology in Jourdan River

For the previous developed Jourdan River watershed model, the simulation period was from January 1, 1965 to May 31, 1999. However, the observed water quality data are only available for water year 2000 and 2001. Without observed data to calibrate the watershed model, the accuracy of the developed water quality model would be unknown. Hence, it was necessary to extend the simulation period of Jourdan River watershed model to water year 2001.

HSPF requires the hourly input meteorological data including precipitation, air temperature, dewpoint temperature, wind movement, solar radiation, cloud cover, potential evapotranspiration and surface evaporation. For the previous developed watershed model, the meteorological data from station MS226921 were applied to all the sub-watersheds in Jourdan River (Fig. 4.1-28). The precipitation data from other five stations around the study area, MS227128, MS229617, MS228352, MS220521, and LA168539, were evaluated to examine which one is more suitable for Jourdan River watershed.

The meteorological data at stations of MS228352 and LA168539 were excluded from further analysis due to the limitation of observed duration. The duration of meteorological data at the station MS228352 is from 1948 to 1988, however, the observed water quality data are only available from 2000 to 2001. The meteorological data at the station LA168539 are from 1974 to present, however, the observed flow data available for calibration are only from 1965 to 1966. For the remaining three meteorological data, linear regression analysis was conducted between the monthly precipitation and streamflows from January 1, 1965 to September 31, 1966.

The precipitation data at station MS227128 have a closer relationship with monthly streamflows than the other two stations (Table 4.1-19). Hence, the meteorological data from station MS227128 were applied to all the sub-watersheds in

Jourdan River, and the comparison of current and previous hydrographs was shown in Fig. 4.1-29. Generally, the previous model (with R^2 of 0.6318) simulated observed data slightly better than current model (with R^2 of 0.5948). However, for the current model, the simulation period was extended from January 1, 1965 to May, 24, 2001. Hence, the observed water quality data in Jourdan River by MDEQ could be used to calibrate the model.

Table 4.1- 19. Linear regression analysis between monthly precipitation and flow.

Meteorological Station ID	MS220521	MS227128	MS229617
R^2	0.3125	0.4366	0.3627

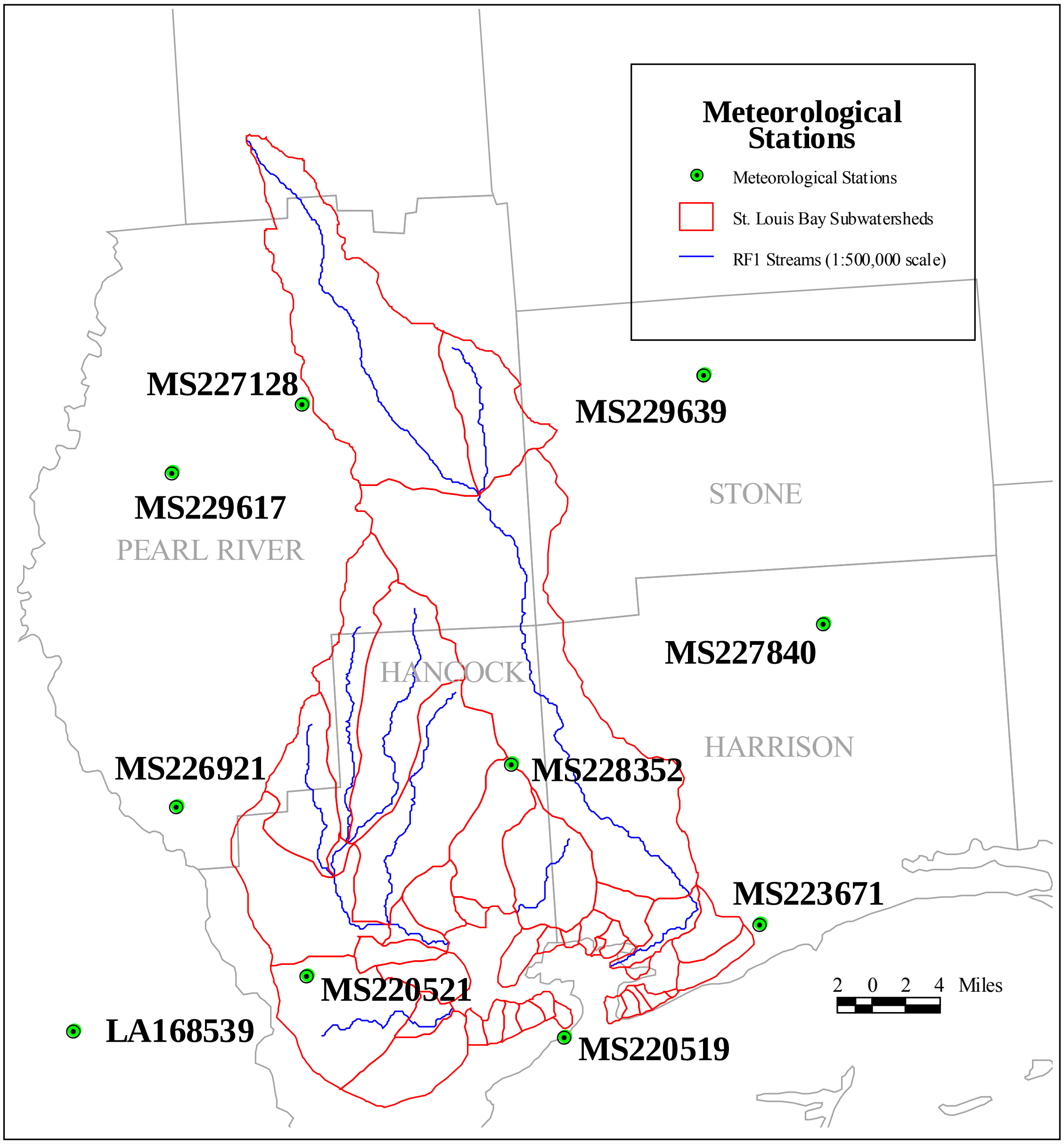


Fig.4.1- 28. Meteorological Stations within or near the St. Louis Bay Watershed

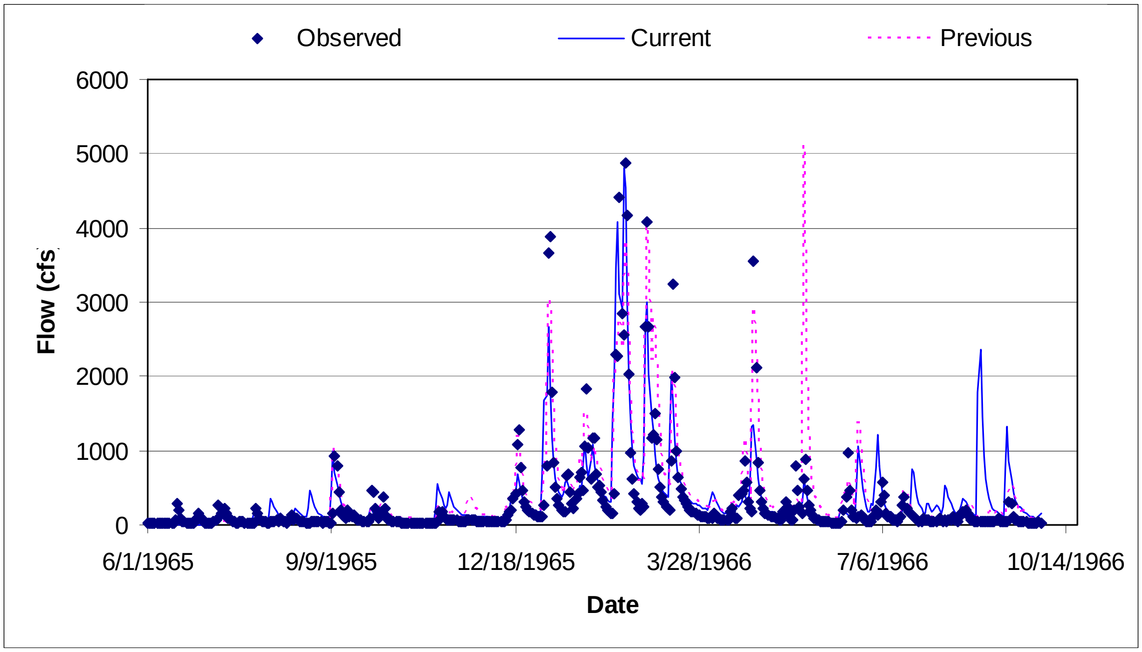


Fig.4.1- 29. Comparison of previous and current hydrographs.

Confirmation of Model Inputs

Some of the developed model inputs have been substantiated by soil sampling and edge-of-field data. The results of soil sampling and edge-of-field experiment were presented separately.

Soil Sampling

The soil samples from St. Louis Bay study area were collected and analyzed. The sampling results from a similar agricultural watershed at Mississippi State University Pontotoc Ridge/Flatwoods Branch Experiment Station (Evans, 2005) were also used to confirm the model inputs.

Soil Sampling in the St. Louis Bay Watershed

The sampling locations were selected in Pearl River, Hancock, and Harrison counties so as to represent the widest possible array of soil types and agricultural land practices. These sites span the study area (Fig.4.2-1). The samples represent these series and the associated soil series. Taken together they represent between 40 to 60 percent of the soils in the Bay St. Louis Study area (USDA-NRCS, 2006). The soil samples were taken from 0 to 1 in., 1 to 6 in, 6 to 18 in., and 18 to 30 in. depths. These depths were chosen to provide the basis for parameter estimation for the various soil depths in the AGCHEM module. Soil nutrient quantities were determined using procedures described by Sparks et al. (1996).

Fig. 4.2-2 and Fig.4.2-3 gave the typical spatial distribution of nitrogen and phosphorus in the measured soil samples from the study area. It can be observed that most of the nutrients in the soil samples are found in the soil depth of 0-6 inches (Fig. 4.2-2 and Fig.4.2-3). In addition, the amount of nutrients in the soil depth of 0-1 inch is much greater than that in the soil depth of 1-6 inch.

The development of nitrogen and phosphorus loadings from fertilization, for the St. Louis Bay watershed model, was documented at the beginning of Chapter 4. The nutrient loadings from hay cropland dominate the nutrient contributions from croplands due to the comparatively larger area and higher unit loading rates. For the developed model, it was assumed that the typical application method is broadcast, which only applies the nutrients to the surface soil layer, with the prescribed soil depth of 0 – 0.5 inch.

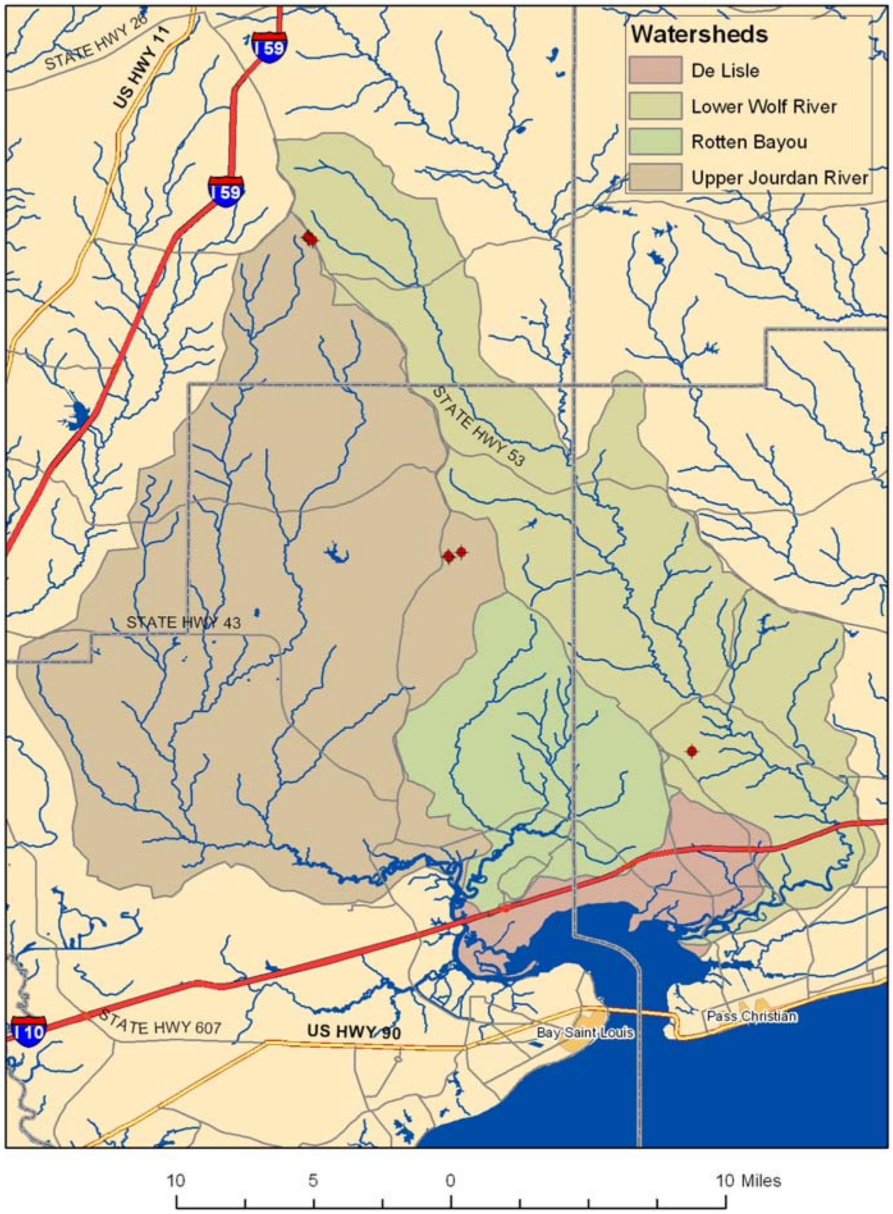


Fig.4.2- 1. Sample locations shown in sub-watersheds of the St. Louis Bay watershed. Samples located in Pearl River, Harrison and Hancock counties (north to south).

The results of nutrient distribution in the soil samples proved the validity of assumed nutrient distribution in the developed model. For the model, all the nutrients from fertilization was assumed to be applied in the top soil layers, which could cause the majority of nutrients to remain in the soil surface layer, which was demonstrated by the nutrient distribution in the soil samples. In addition, the phosphorus transportation with vertical water flow could cause the stepwise-deceasing of phosphorus level in the vertical soil profile, which was also demonstrated by the nutrient distribution in the soil samples.

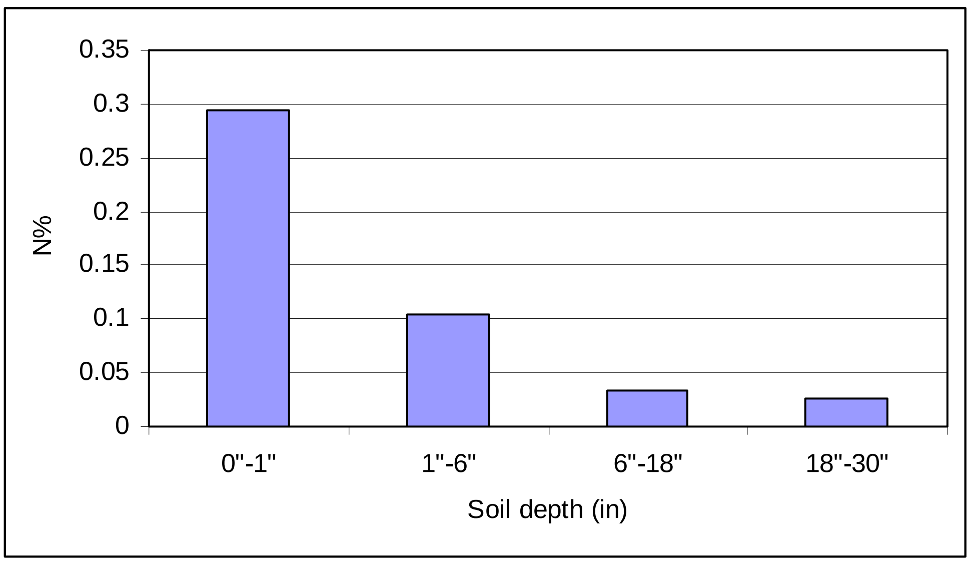


Fig.4.2- 2. Typical spatial distribution of nitrogen in the soil.

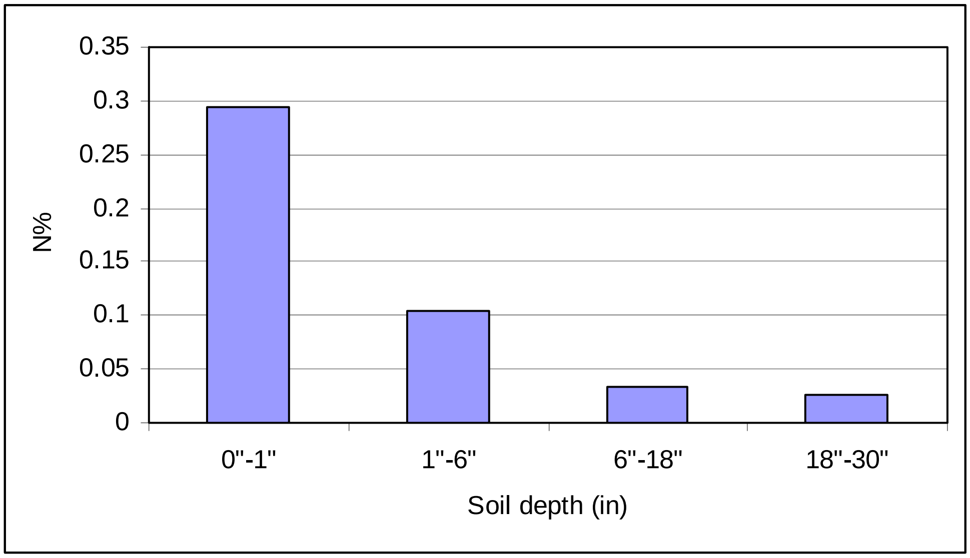


Fig.4.2- 3. Typical spatial distribution of phosphorus in the soil.

Soil Sampling in the MSU Pontotoc Ridge/Flatwoods Branch Experiment Station

Evans (2005) investigated the relationships among phosphorus concentrations and soil properties and land use in a 259-acre agriculture watershed on the Mississippi State University Pontotoc Ridge/Flatwoods Branch Experiment Station. Over the entire study area, 400 soil samples were collected and analyzed for PH, Mehlich Ш-extractable P (M3P), Olsen-extractable P, and total P.

The spatial distribution of mean M3P in the soil was shown in Fig.4.2-4. The soil samples were collected from four layers: thatch, 0-3 inch, 3-9 inch, and 9-18 inch. In order to be compared with model inputs, the spatial distribution of M3P was converted to be compatible with soil layers specified in our model: 0-0.5 inch, 0.5-6.5 inch, and 6.5-47.5 inch. In order to do this, several assumptions have to be made. The measured M3P in each soil layer is assumed to be uniformly distributed over the entire soil layer. The measured M3P in the Thatch is assumed to be in the soil surface layer specified by the model since Thatch is the interface between vegetation and soil. The adjusted spatial distribution of M3P was shown in Fig.4.2-5. It can be observed that the result also proved the validity of assumed nutrient distribution in the developed model, which assumed that all the phosphorus from fertilization was assumed to be applied in the top soil layers.

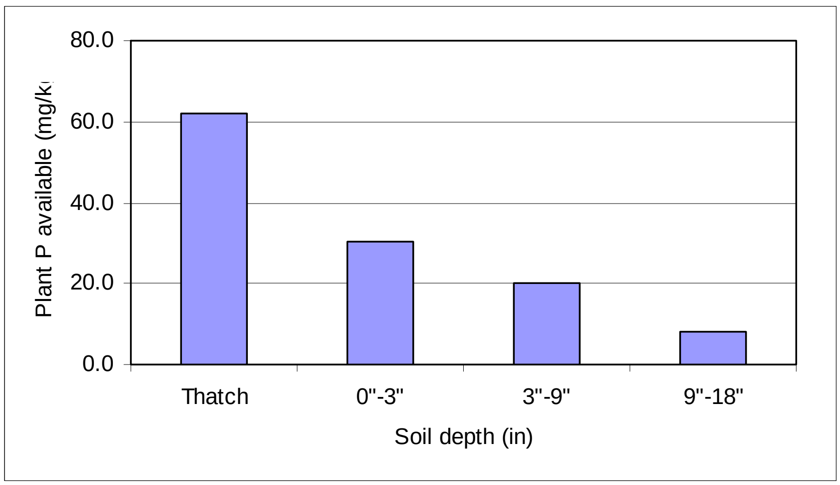


Fig.4.2- 4. Measured mean M3P distribution in the soil.

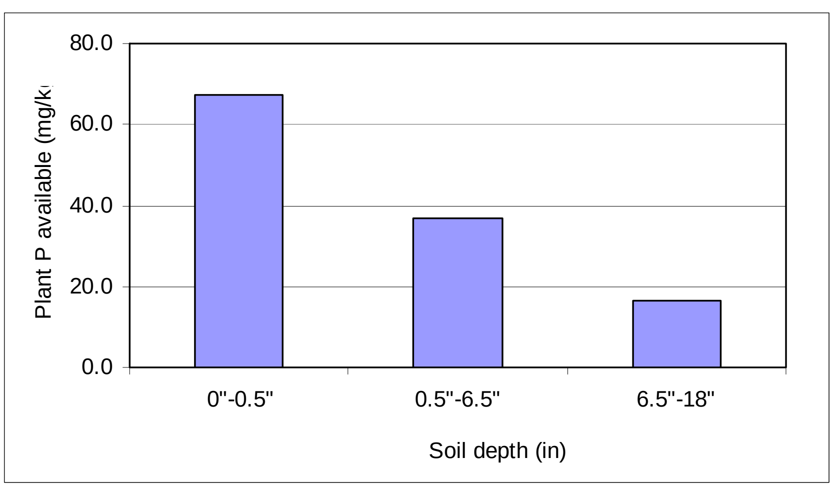


Fig.4.2- 5. Adjusted M3P distribution in the soil.

Edge-of-field Experiment

Beavers (2005) initiated the edge-of-field experiment to evaluate the dynamics and forms of phosphorus in sediment and runoff, determine phosphorus losses under two tillage and two planting treatments, and examine phosphorus concentrations resulting from rainfall and runoff influence. This study was conducted at the North Mississippi Branch of the Mississippi Agriculture and Forestry Experiment Station (MAFES) in Holly Springs, Mississippi, in conjunction with the USDA-ARS National Sedimentation Laboratory in Oxford, Mississippi. The applied phosphorus fertilizer was chicken litter. Edge-of-field data also confirmed the nutrient inputs for the developed model. For the developed St. Louis Bay watershed model, the phosphorus input from fertilization practice was assumed to be 100% in the form of inorganic phosphorus, PO_4. This assumption has been substantiated by the edge-of filed data. The majority of phosphorus in the surface runoff was in the form of inorganic phosphorus (Fig.4.2-6) (Beaver, 2005). The nature of phosphorus in poultry litter is like phosphorus fertilizers.

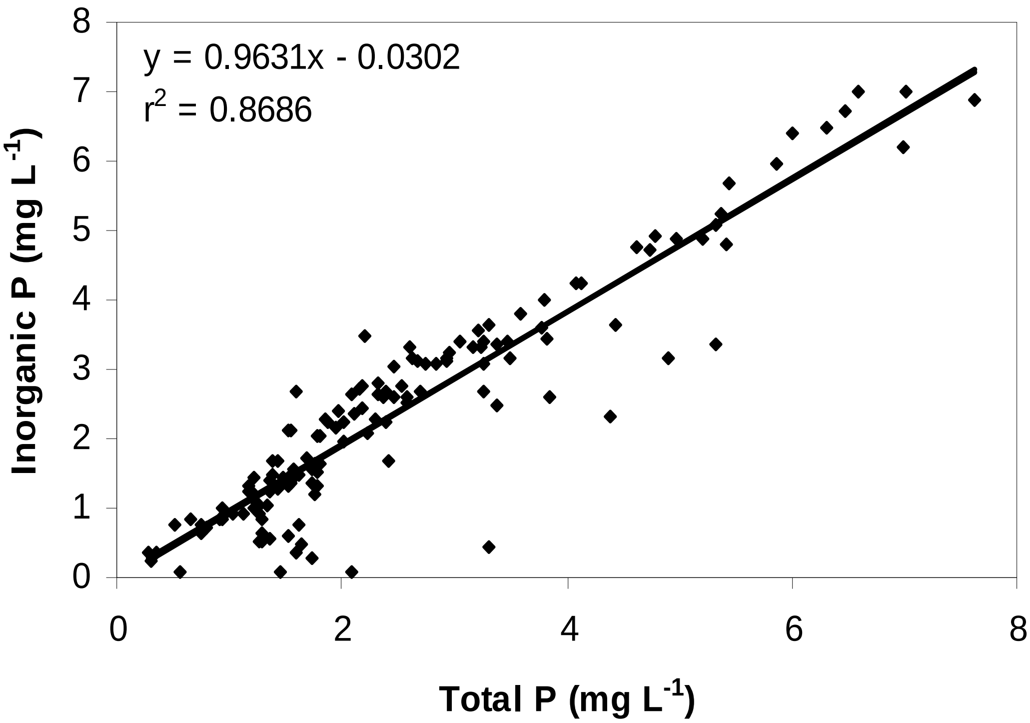


Fig.4.2- 6. Phosphorus concentration (mg L^{-1}) for three treatments and daily rainfall amounts for a 6 month period in the Lexington sites (After Beaver, 2005).

The developed phosphorus mass balance in the hay cropland for the St. Louis Bay watershed model was also substantiated by the edge-of-field experiment. The results from edge-of-field experiment indicated that the ratio of phosphorus uptake to phosphorus input ranged from 6.15% to 9.82%, whereas the ratio in the St. Louis Bay model was 8.69% (Table 4.2-1). It can be observed that the phosphorus input for the St. Louis Bay model is lower than that by Beavers (2005). This is because the phosphorus rate used in the St. Louis By model reflects the average fertilization condition over the entire simulation period from 1965 to 2001, whereas the loading rate from Beavers (2005) represents the loading rate of recent decade. In both cases, the phosphorus application rates were developed based on nitrogen application rate, which could cause the higher levels of phosphorus in the surface runoff and in-stream.

Table 4.2- 1. Comparison of phosphorus mass balance between edge-of-field experiment and St. Louis Bay model inputs.

Year	P input	P uptake	Uptake: Input
Fall 1997 (Beaver, 2005)	163.91	10.08	6.15
Spring 1998 (Beaver, 2005)	174.76	10.87	6.22
Spring 1999 (Beaver, 2005)	133.12	13.08	9.82
Spring 2000 (Beaver, 2005)	176.69	17.42	9.71
St. Louis Bay model	101.30	8.80	8.69

CHAPTER V

CALIBRATION OF WATER QUALITY MODEL

Several sources of in-stream monitoring data are available for calibrating the watershed water quality model. Historical observed data from USGS gauge station on Wolf River near Landon, Mississippi (USGS Station 02481510) were one data source for calibration. Another data source is from Mississippi Department of Environmental Quality (MDEQ). From water year 2000 to 2001, MDEQ collected water quality data during 8 storm events and 4 base flow events to assist in the development of a watershed model. The St. Louis Bay watershed water quality model was initially calibrated by using the observed data of USGS gauge station 02481510 and MDEQ sampling station WR2 (Kieffer, 2002 and Huddleston et al., 2003). The USGS gauge station 02481510 and MDEQ sampling station WR2 are located at the same site. The modeling scenario 2 was developed by using the nutrient loading scenario 2 from fertilization practice, adjusted plant uptake, calibrated BOD decay rate, and site-specific pollutant accumulation rates from non-cropland, and recalibrated hydrology in Jourdan River. The calibration results of modeling scenario 2 were compared with the initial calibration results, which was referred as modeling scenario 1. In addition, the calibration was extended to Jourdan River by using the observed data from MDEQ sampling station JR3. The nutrient calibration results were also compared with the observed data from the USGS gauge station 02481510. The simulation of water temperature was also included herein since it is very crucial for the DO simulation.

Water Temperature Simulation

The simulation of water temperature was evaluated using the USGS observed data from 1978 to 1986 (Fig. 5.1-1). The water temperature was observed monthly from 1978 to 1980, bi-monthly from 1981 to 1982, and quarterly from 1983 to 1986. The simulated water temperature closely matched the observed data and correctly captured the seasonal change of water temperature (Fig. 5.1-1). Higher value of determination coefficient (R^2) indicated that the majority of variations in the observed water temperature have been explained by the simulated water temperature (Fig. 5.1-2).

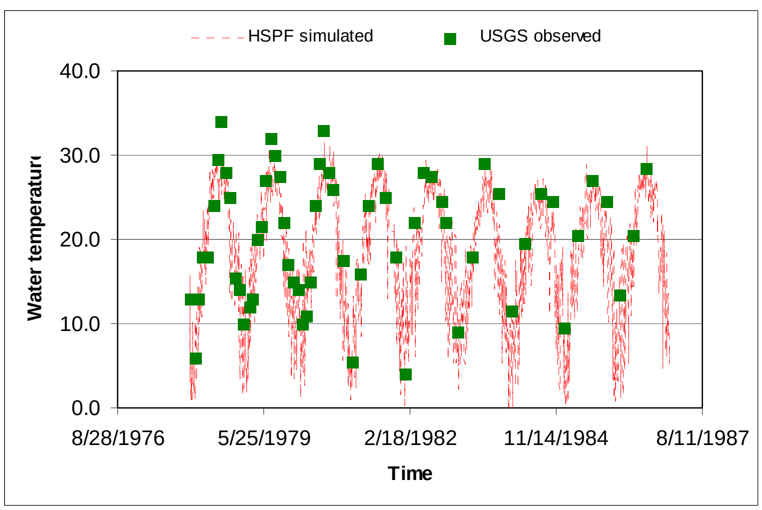


Fig.5.1- 1. Simulation of water temperature at USGS station 02481510

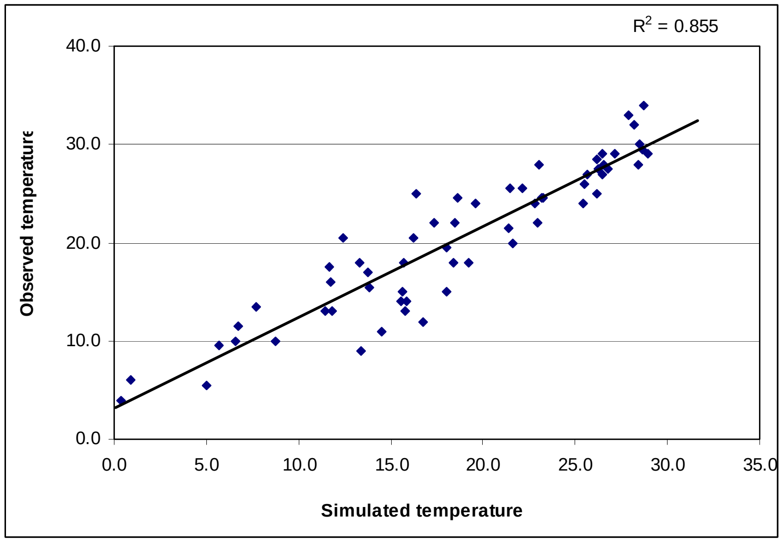


Fig.5.1- 2. Scatter plot of simulated temperature against observed temperature

The model simulation of water temperature was also compared with the MDEQ observed data from 2000 to 2001 at the sampling station WR2 and JR3 (Fig. 5.1-3 and 5.1-4). Generally, the modeling performances at the sampling station WR2 and JR3 were good and the simulated water temperature reflected the overall temporal trend of observed data (Fig. 5.1-3 and 5.1-4). However, the calculated values of R^2 for WR2 and JR3 were 0.302 and 0.347, respectively, much lower than 0.855 for USGS gauge station 02481510. The differences of modeling performance in water temperature simulation could be caused by the deficiencies of HSPF in modeling water temperature. Chen et al. (1998a and 1998b) pointed out several deficiencies of HSPF in simulating soil temperature and water temperature; firstly HSFP can not take onto account for the impacts of vegetation cover on soil and outflow temperature and secondly HSPF can not represent the dynamic shading impacts of riparian vegetation and topography on water temperature simulation. The difference in modeling performance of water temperature between 1978 to 1986 and 2000 to 2001 could be caused by the land use change in the study area. The land use change has strong effects on watershed cover vegetation and riparian vegetation, which, in turn, exerts strong impacts on soil water temperature and water temperature. Correcting such a deficiency would require both a more extensive modeling effort and more detailed field data for calibration purposes.

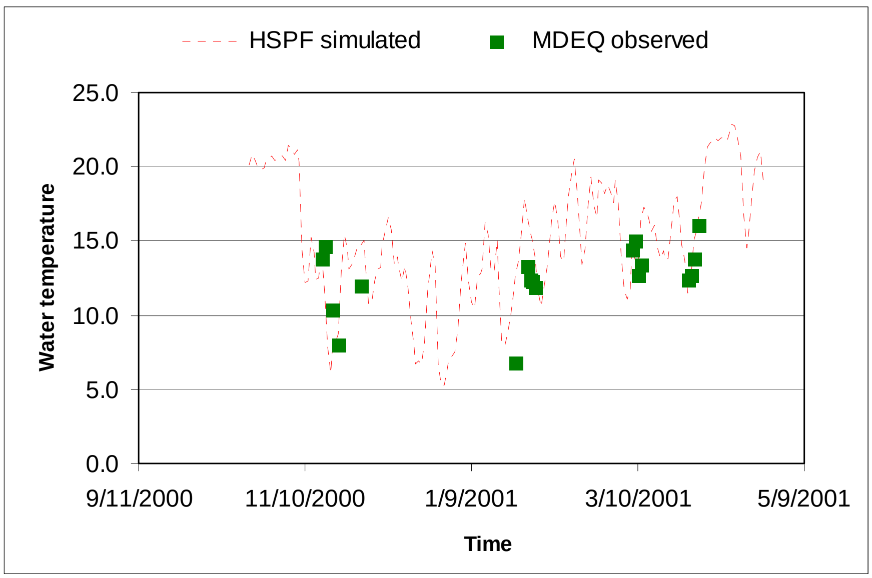


Fig.5.1- 3. Simulation of water temperature at MDEQ station WR2

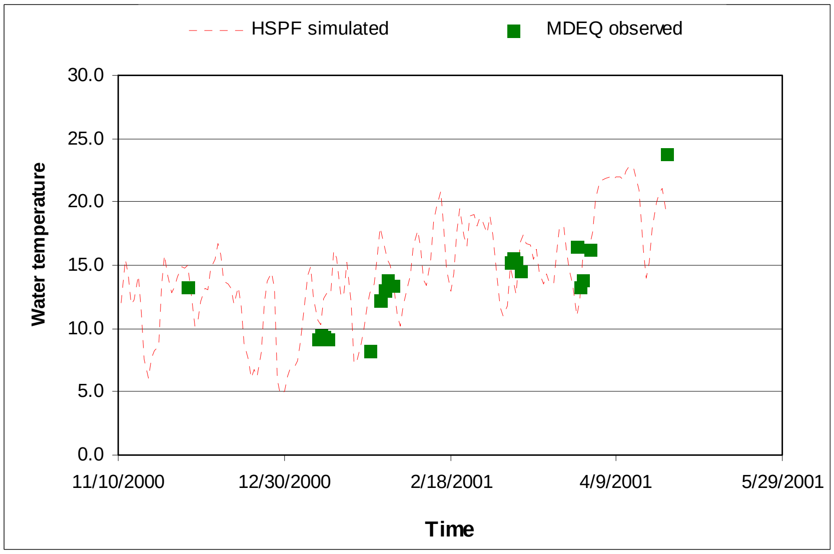


Fig.5.1- 4. Simulation of water temperature at MDEQ station JR3

DO Calibration

For modeling scenario 1, the simulation period was from January 1, 1978 to December 31, 1980, whereas the simulation period was extended to December 31, 1986 for modeling scenario 2. For modeling scenario 1, there were several events with simulated DO of zero or approaching zero when calibrating the USGS observed data, which do not represent the realistic conditions (Fig. 5.2-1). The fluctuations of the simulated DO concentration to zero or near zero levels were originally assumed to be a result of deficiencies of the hydrology model relative to extreme low flow events. Because the hydraulic simulation is based on uniform stream geometry, extreme low flow events result in model calculation of very shallow stream depths. With very shallow stream depths, the simulated water temperature greatly increases, resulting in a magnified misrepresentation of dissolved oxygen consumption in the system (Kieffer, 2002).

However, the simulated water temperature ranged from 0.1 to 31.1˚C, and there were no unrealistically higher values of simulated water temperature (Fig. 5.1-1).

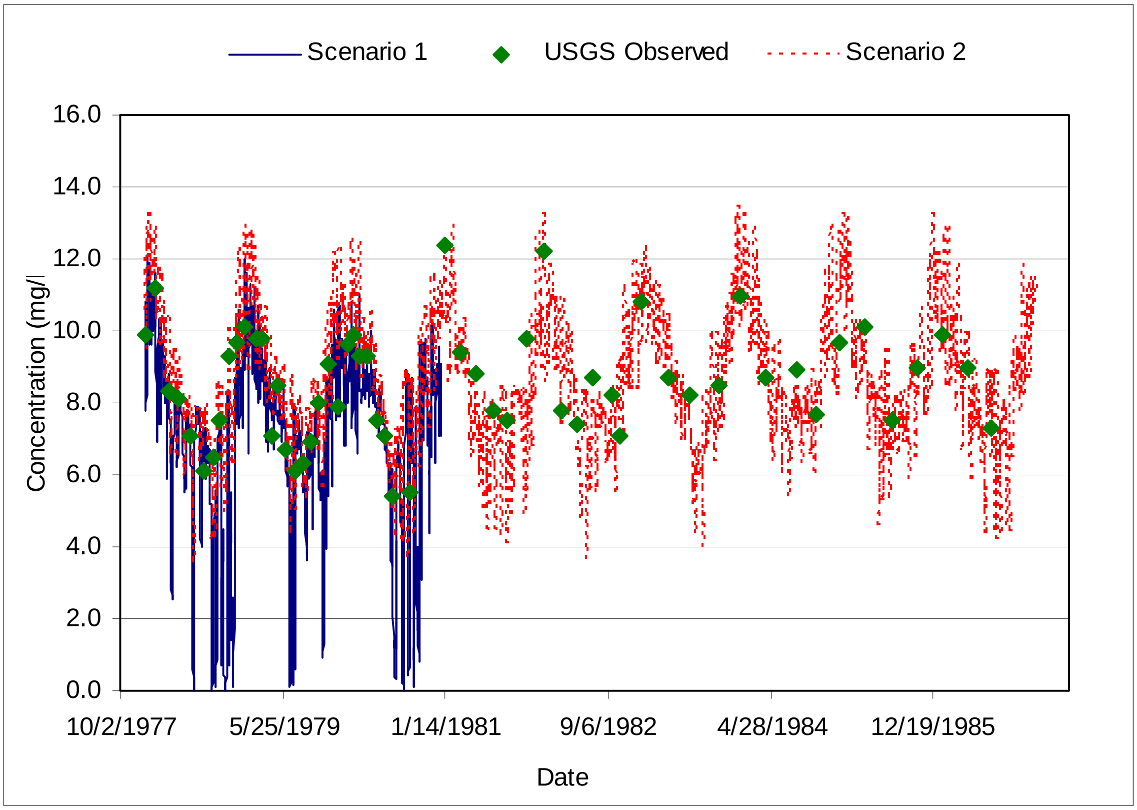


Fig.5.2- 1. Simulated DO at USGS 02481510 for modeling scenarios 1 and 2

During the calibration processes, it was found that BOD decay rate was a very sensitive parameter. The fluctuations of the simulated DO concentration to zero or near zero levels in modeling scenario 1 were caused by using higher value of BOD decay rate. For modeling scenario 2, the BOD decay rate was reduced from 0.05 hr^{-1} used in modeling scenario 1 to 0.005 hr^{-1}, remaining within the range of recommended rates of BOD decay. The value of 0.004/hour of BOD decay rate was used by Donigian et al

(1994) to calculate the nutrients loadings from Chesapeake Bay watershed. For modeling scenario 2, the simulation results of DO in the USGS gauge station closely match the observed data and follow the sinusoidal pattern of the monthly-monitored DO concentrations, with lower dissolved oxygen supply in the summer months when high temperatures lead to higher growth and oxygen consumption from aquatic populations (Fig. 5.2-1).

The modeling performances of DO at sampling station of WR2 and JR3 were not as good as at USGS station 02481570. The in-stream concentrations of DO were often over-predicted at sampling station of WR2 and JR3 (Fig. 5.2-2 and Fig.5.2-3). Through analysis of the observed data, it was found that the observed concentrations were highly correlated with observed water temperature at the sampling station WR2 and JR3 (Fig. 5.2-4). The discrepancies between the observed and simulated DO could be caused by the water temperature simulation. To prove this, the errors between observed and simulated DO, and the errors between observed and simulated water temperature were calculated, then regression analysis was conduct between these two errors. There was strong negative linear relationship between these two errors (Fig. 5.2-5). The negative linear relationship means that once the water temperature is under-estimated, the DO will be over-predicted, and vice versa. In addition, high value of R^2 give evidence that the majority of the variations in the errors between observed and simulated DO is explained by the errors of water temperature simulation.

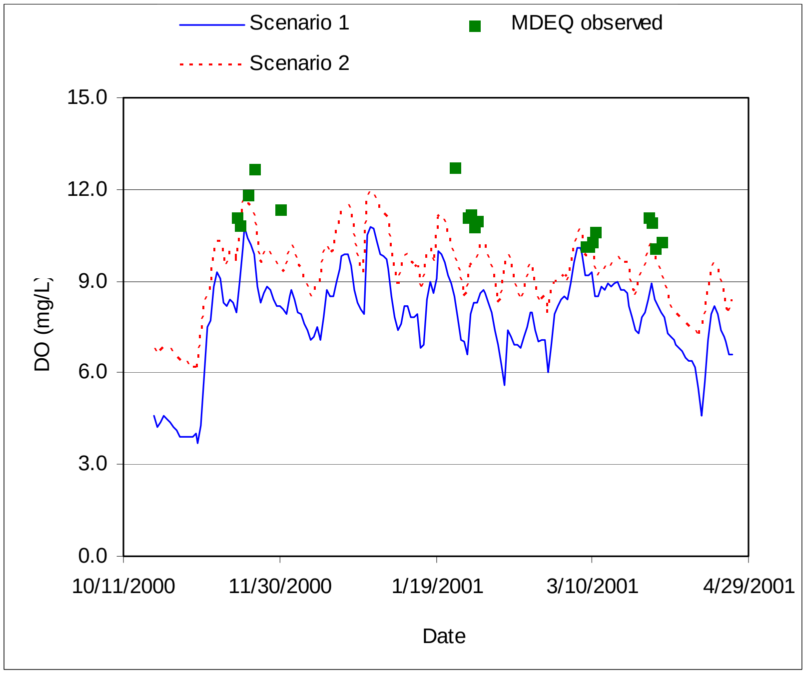


Fig.5.2- 2. Simulated DO at MDEQ station WR2 for modeling scenarios 1 and 2

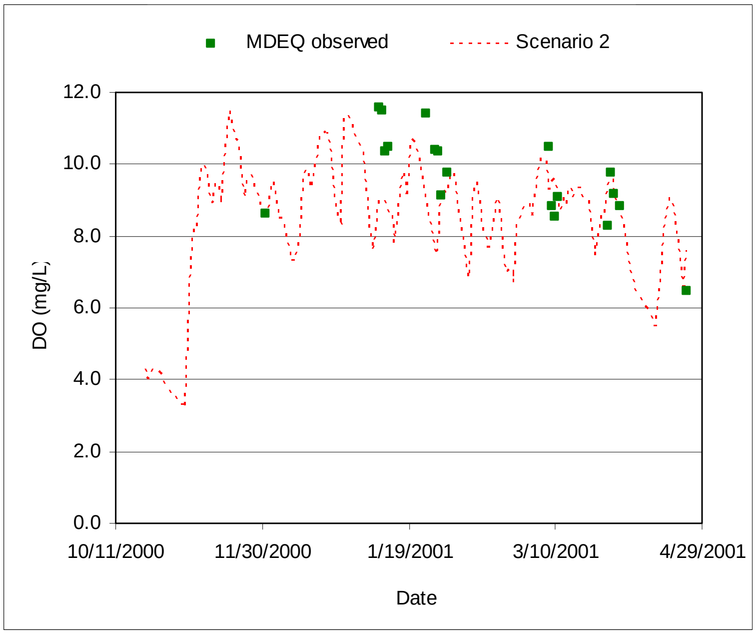


Fig.5.2- 3. Simulated DO at MDEQ station JR3 for modeling scenario 2

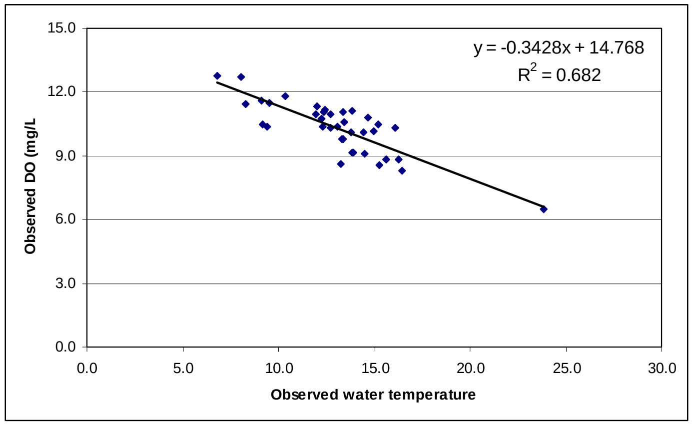


Fig.5.2- 4. Linear regression analysis between observed DO and observed water temperature

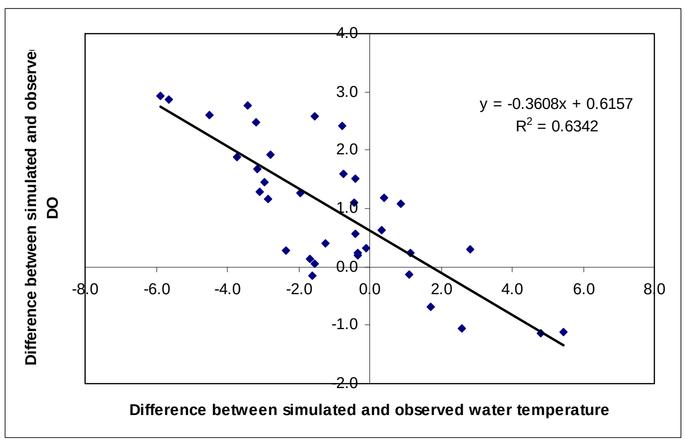


Fig.5.2- 5. Linear regression analysis between simulated errors in DO and simulated errors in water temperature.

It was also found that nitrification and denitrification processes have very slight effects on DO simulation, which could be attributed to the low level of in-stream NO_3 and NH_4 concentrations. The average values of observed NO_3 and NH_4 concentrations are 0.056 and 0.012 mgL^{-1}, respectively at the sampling station WR2, and 0.02 and 0.02 mgL^{-1}, respectively at sampling station JR3.

BOD Calibration

There was no USGS observed BOD available to calibrate the BOD, and BOD simulation was only evaluated at the MDEQ sampling station WR2 and JR3 (Fig. 5.3-1 and Fig.5.3-2). For modeling scenario 2, the parameter BROBOD1, benthal release rate of BOD at high oxygen concentration, was adjusted so that the contributions of nutrients, such as NH_4 and PO_4, from BOD decay would not exceed the observed levels. Finally, the value of BROBOD1 was adjusted from 72 $mgm^{-2}hr^{-1}$, used in modeling scenario 1, to 18 $mgm^{-2}hr^{-1}$. The modeling performances of BOD were nearly same for modeling scenarios 1 and 2. The simulated BOD concentrations closely match the observed data at both sampling stations. At the sampling station of JR3, the observed value of BOD on January 25, 2001 was 11.87 mgL^{-1}, approximately 5 times higher than the average value of the observed BOD (Fig. 5.3-2). The developed model did not capture this extreme event, but represent the general trend and average condition of observed BOD very well (Fig. 5.3-2). The unusually higher value of observed BOD could be related to observation errors or other accident factors.

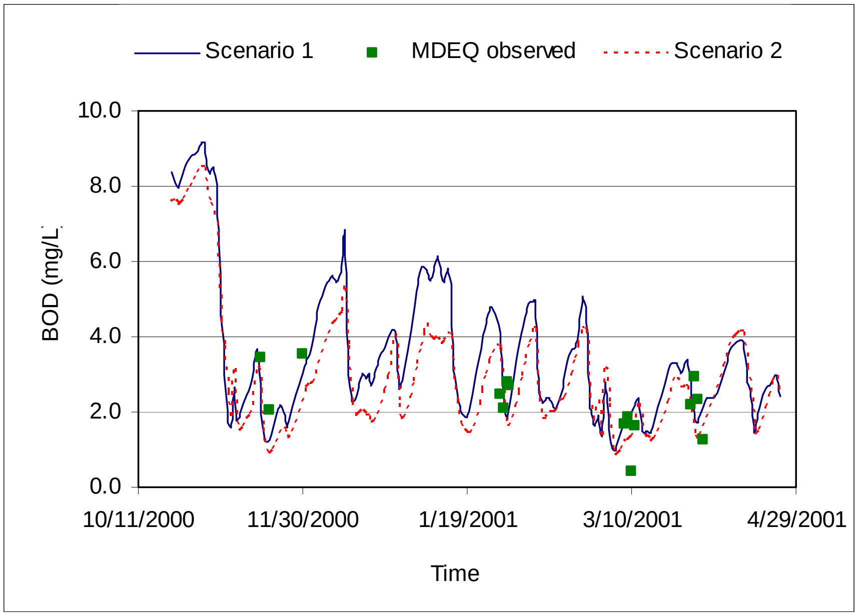


Fig.5.3- 1. Simulated BOD at WR2 for modeling scenarios 1 and 2

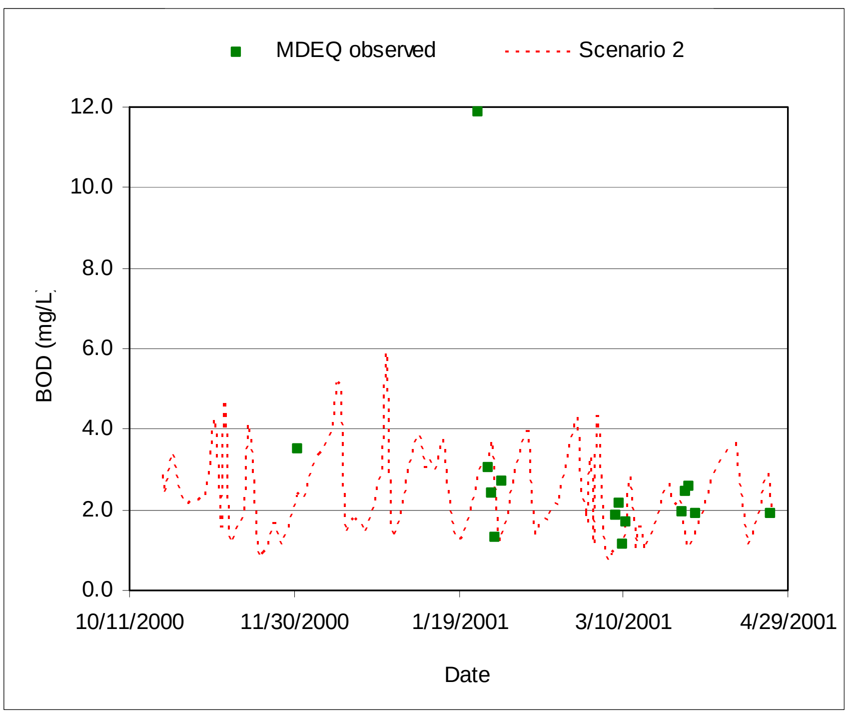


Fig.5.3- 2. Simulated BOD at JR3 for modeling scenario 2

Nutrient calibration

For modeling scenario 2, the developed loading scenario 2 for fertilization practice was used as the model input of nutrient. The nutrient input from other sources including atmospheric deposition, NPDES sources, failing septic system, and manure application were same as modeling scenario 1. Since the preparation of nutrient input using Manual Time Series method involves in too much time and efforts, Monthly Data Block method were used for both modeling scenarios.

For both modeling scenario 1 and 2, Yield-based algorithm was used to simulate plant uptake process. The total annual plant uptake target was developed by based on the annual average crop yields and nutrient composition in the dry weight. For modeling scenario 1, the annual average yield in recent decade, 1990-2000, was used to calculate the plant uptake target, whereas the annual average yield over the entire simulation period was used for the modeling scenario 2. The monthly fraction of annul uptake target for corn was modified based on the studies of Iowa State University for modeling scenario 1. The monthly fraction of annual uptake target for other croplands and the soil layer fraction of monthly uptake rate for all croplands were same for scenario 1 and 2.

As stated before, the total annual uptake target is not the intended amount of nutrient removed by the plant uptake, but affected by the soil nutrient and moisture level. Hence, the trial-and-error method was used to calibrate the annual plant uptake target input to the intended amount. During the calibration process of annual plant uptake rate for nitrogen, it was found that the model was not able to generate the intended amount of nitrogen even with unrealistically higher value of annual plant uptake rate for hay and corn croplands. The generated average annual plant uptake rates with input uptake rate of

5,000 lb/ac were 45.5 and 26.3, respectively for hay and corn croplands, less than the intended uptake rates of 63.0 and 50.7 lb/ac. Much higher value of uptake target was tried, but the generated uptake rates were still less than the intended rates. After examining the nitrogen input and plant uptake, it was found that the nitrogen input was assumed to be 25% in the form of NO_3 and 75% in the form of NH_4, and the nitrogen uptake ratio of NO_3 and NH_4 was 1: 0 for all croplands. For hay cropland, the nitrogen input from fertilization was 101.4 lb/ac, 25.35 lb/ac input of NO_3 and 76.05 lb/ac input of NH_4. However, the annual nitrogen uptake rate for nitrogen was 63.0 lb/ac, completely in the form of NO_3. For the annual nitrogen balance, the uptake rate of NO_3 is too much higher than the amount of NO_3 available in the soil, which is the reason why so much higher input value of nitrogen uptake rate still could not generate the intended amount of uptake. The same problem happened in the corn cropland. Since the adjustment of nitrogen input takes much time and efforts than changing the uptake ratio of NO_3 to NH_4, it was determined to modify the uptake ratios of nitrogen to satisfy the intended nutrient mass balance. The uptake ratio of NO_3 and NH_4 was modified from 1: 0 to 0.25: 0.75 for modeling scenario 2. With the new input of uptake ratio of NO_3 and NH_4, the model was able to generate the intended nitrogen uptake rates. The intended annual plant uptake rate, generated plant uptake rate, and calibrated plant uptake rate for modeling scenario 2 were given by Table 5.4-1. It can be observed that the generated amount of annual plant uptake was lower than the intended plant uptake rate, especially for nitrogen, the generated uptake rates were less than half of the intended uptake rates.

Table 5.4- 1. Intended and calibrated total annual plant uptake target for croplands.

Cropland	Intended N uptake (lb/ac)	Generated N uptake (lb/ac)	Calibrated N uptake (lb/ac)	Intended P uptake (lb/ac)	Generated P uptake (lb/ac)	Calibrated P uptake (lb/ac)
Wheat	38.6	18.1	200	4.8	4.5	5.05
Corn	50.7	22.2	1,745	8.8	8.6	8.95
Soybean	0.0	0.0	0.0	8.8	8.5	9.5
Hay	63.0	22.9	215	6.4	6.3	6.5

For modeling scenario 2, values of ACQOP for non-croplands were calibrated to match the observed data in Wolf and Jourdan River. During the calibration processes, the values of ACQOP for non-croplands were simultaneously increased or decreased by keeping the relative ratios of ACQOP among land-uses constant. The calibrated values of ACQOP for Wolf River and Jourdan River were shown in Table 5.4-2.

Table 5.4- 2. The developed ACQOP for modeling scenario 1 and 2.

ACQOP (lb/ac-year)		NO_3		PO_4	
		Scenario 1	Scenario 2	Scenario 1	Scenario 2
Wolf River	Pasture	5.18	0.32	1.82	10.92
	Forest	0.89	0.06	0.049	0.29
	Upland Scrub/Shrub	0.89	0.06	0.049	0.29
	Pervious Urban	1.64	0.10	0.16	0.96
	Impervious Urban	7.39	0.46	0.74	4.44
	Wetland	2.65	0.17	0.43	2.58
Jourdan River	Pasture	5.18	0.26	1.82	5.46
	Forest	0.89	0.04	0.049	0.15
	Upland Scrub/Shrub	0.89	0.04	0.049	0.15
	Pervious Urban	1.64	0.08	0.16	0.48
	Impervious Urban	7.39	0.37	0.74	2.22
	Wetland	2.65	0.13	0.43	1.29

Fig.5.4-1 to Fig.5.4-10 display the simulation results of nutrient at these three stations. The comparisons of modeling performance of NO_3, NH_4, and PO_4 at MEDQ station WR2 between modeling scenarios 1 and 2 were shown in Fig. 5.4-1, Fig. 5.4-4, and Fig. 5.4-7, respectively. For modeling scenario 1, the simulated nutrient concentrations were one or two orders higher than the observed data, whereas the simulated nutrient concentrations were fluctuating in the range of observed data and reflected the general trend of observed data for modeling scenario 1. The majority of modeling performance improvement was attributed to modifying the wrong input unit of nitrogen in Monthly Data Block. In the HSPF manure, the nutrient input unit for Monthly Data Block is lb/ac-month. By examining the code, it was found that the actual input unit should be lb/ac-day. The modeling performance improvement could also attributed to using the more representative nutrient input from fertilization practice, calibrating the uptake-related parameters including total annual uptake target, monthly fraction of annual uptake target, and soil layer fraction of monthly uptake to characterize the nitrogen mass balance, decreasing the BOD decay rate, and calibrating the nutrient accumulation rates for non-croplands.

For modeling scenario 2, the simulation results of NO_3 and NH_4 at USGS gauge station 02481510 and MDEQ station JR3 were shown in the Fig.5.4-2, Fig.5.4-3, Fig.5.4-5 and Fig.5.4-6. The simulated NO_3 and NH_4 concentrations agree fairly well with the observed data; however, there were some unrealistic spikes of simulated nutrient concentrations (Fig.5.4-5). The reasons that caused these spikes will be explored by conducting contribution analysis of source pollutants in the next section.

For modeling scenario 2, Fig.5.4-7 and Fig.5.4-10 display the simulation results of PO_4 at MDEQ station WR2 and JR3. The simulated PO_4 concentrations matched the observed data fairly well, but there were still some high spikes (Fig.5.4-7 and Fig.5.4-10). There were higher variances in the simulated PO_4 than the observed data (Fig.5.4-7 and Fig.5.4-10). The simulation results of PO_4 at USGS gauge station 02481510 were shown in Fig.5.4-8. Generally, the observed PO_4 were constantly over-estimated. There were some very high spikes, such as the simulated PO_4 of 5.43 mg/L on October 17, 1981 and the simulated PO_4 of 4.6 mg/L on July 30, 1986 (Fig.5.4-8). Much of the differences in PO_4 modeling performances between USGS gauge station 02481510 and MDEQ station WR2 were caused by the differences in the magnitudes of the observed PO_4 data between the two stations. USGS gauge station 02481510 and MDEQ station WR2 were located at the same site. The observed PO_4 data used for calibration were from 1978 to 1986 for USGS gauge station 02481510 and from 2000 to 2001 for MEDQ station WR2. The observed PO_4 concentrations in the 1980s were much lower those observed in the early 2000s; the median values of the observed PO_4 were 0.06 and 0.26 mg/L, respectively for USGS station 02481510 and MDEQ station WR2 (Fig.5.4-11). The reasons that caused the increase of observed PO_4 over the time were unknown. Since the PO_4 data were observed by two different agencies, USGS and MDEQ, and was monitored in two different historical periods, part of the differences in the observed PO_4 could be caused by the different monitoring methods. The landuse change also could be the reason that caused the increase in the observed PO_4 over the time. For example, the increase in area of cropland, pasture, and urban land, and decrease in area of forest land could increase the PO_4 loadings from the land surface to the stream, and cause the increase of PO_4 level.

Enough information is not available to identify the accuracies of the observed PO_4 data, and the effects of landuse. In the future studies, it is recommended that more efforts be focused on the impacts of landuse change on PO_4 simulation.

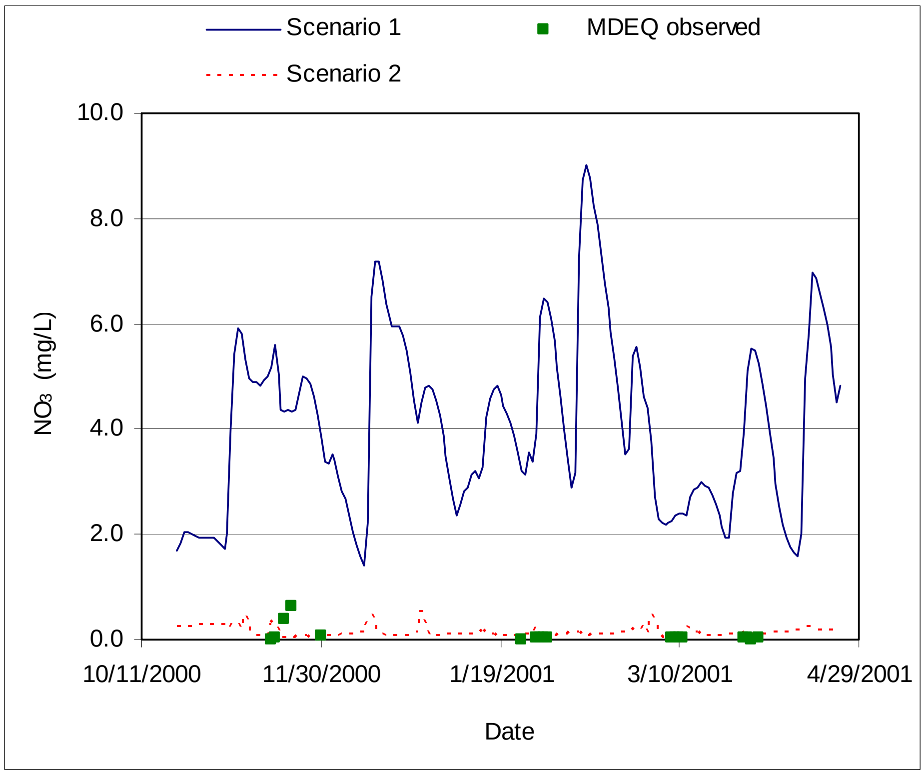


Fig.5.4- 1. Simulated NO_3 at WR2 for modeling scenarios 1 and 2

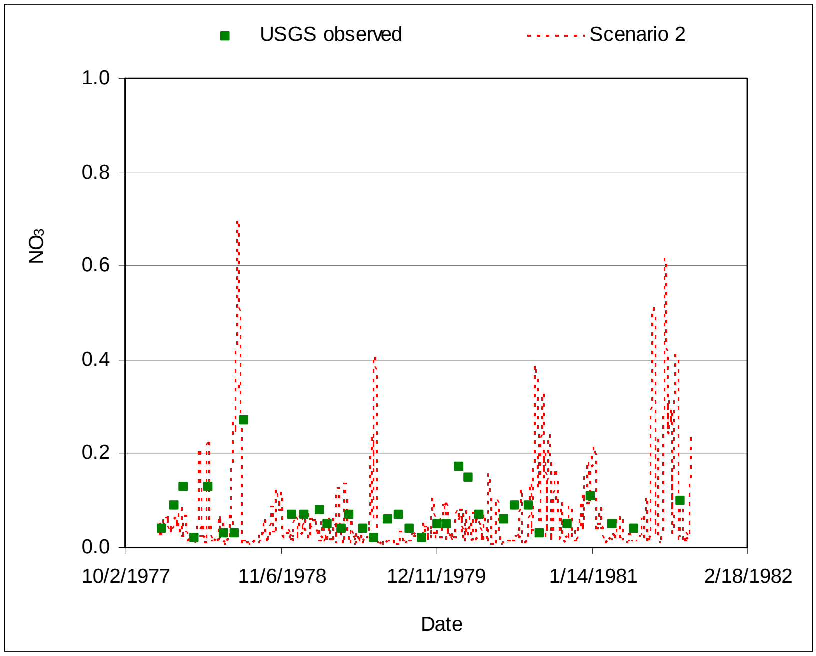


Fig.5.4- 2. Simulated NO_3 at USGS 02481510 for modeling scenario 2

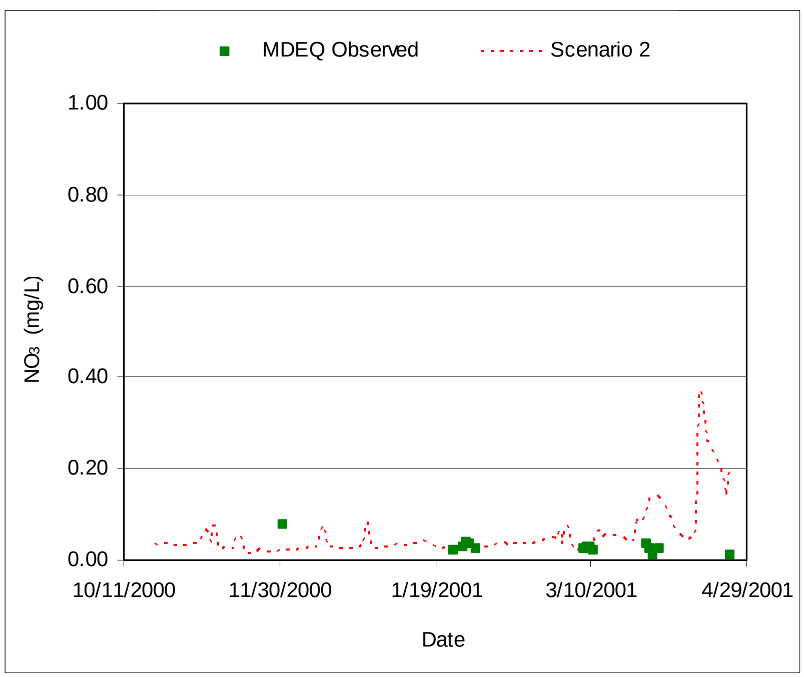


Fig.5.4- 3. Simulated NO_3 at JR3 for modeling scenario 2

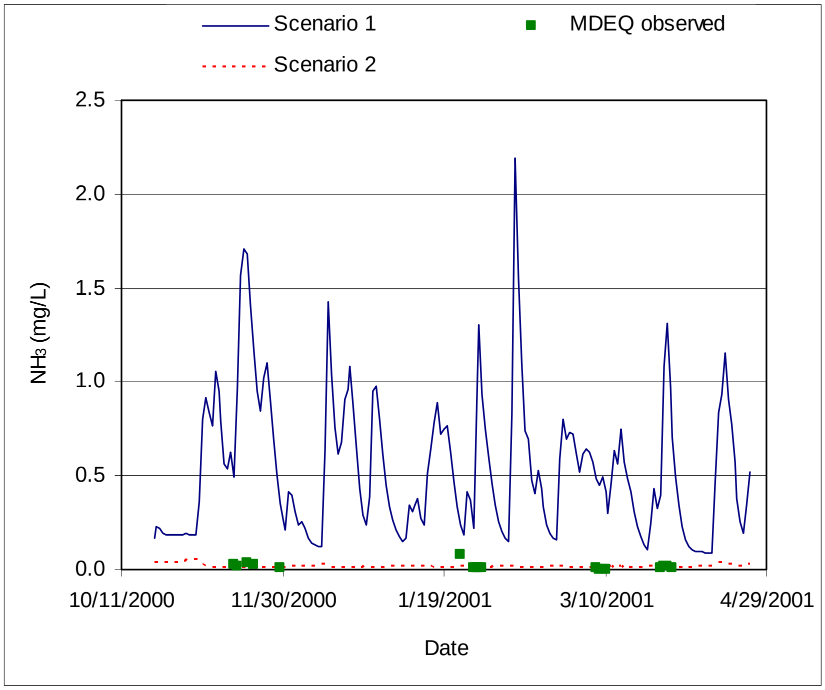


Fig.5.4- 4. Simulated NH_4 at WR2 for modeling scenarios 1 and 2

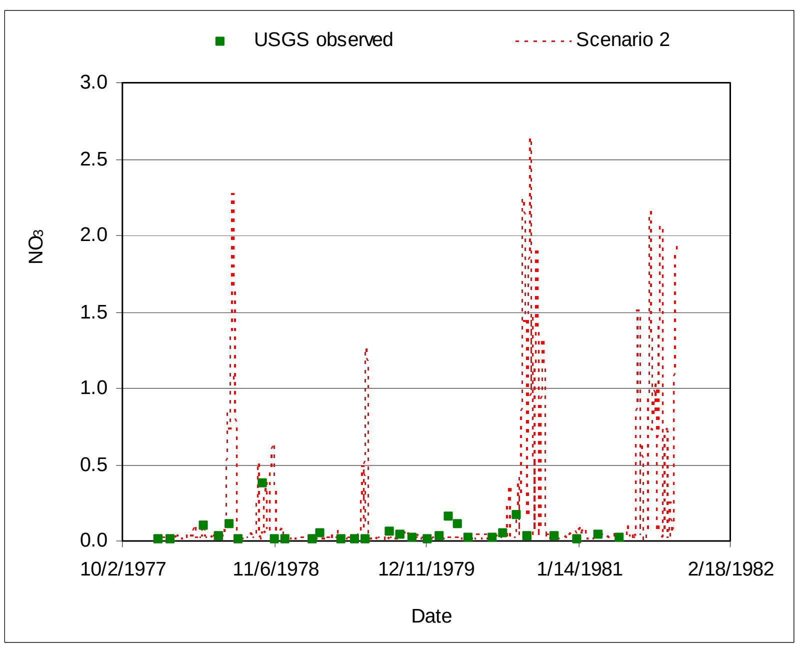


Fig.5.4- 5. SimulatedNH$_4$ at USGS 02481510 for modeling scenario 2

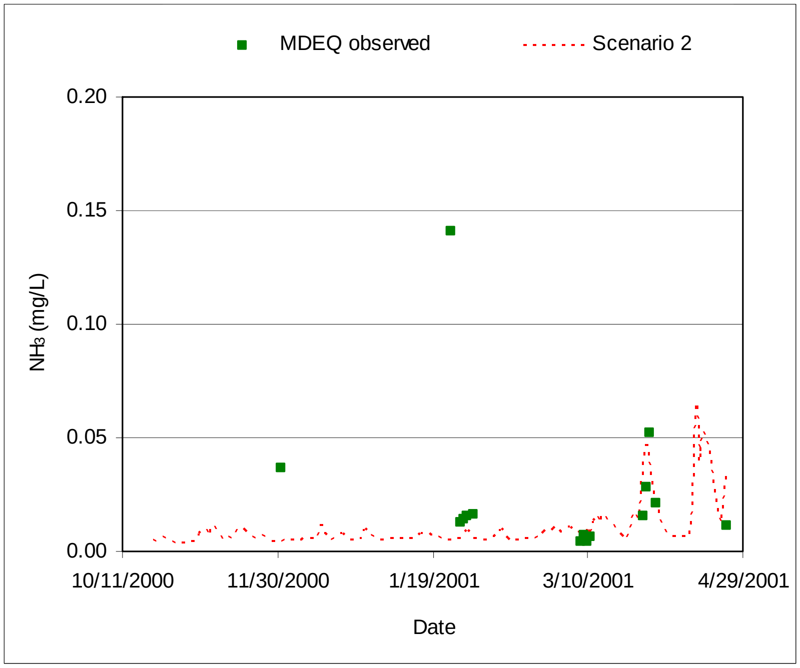


Fig.5.4- 6. Simulated NH_4 at JR3 for modeling scenario 2

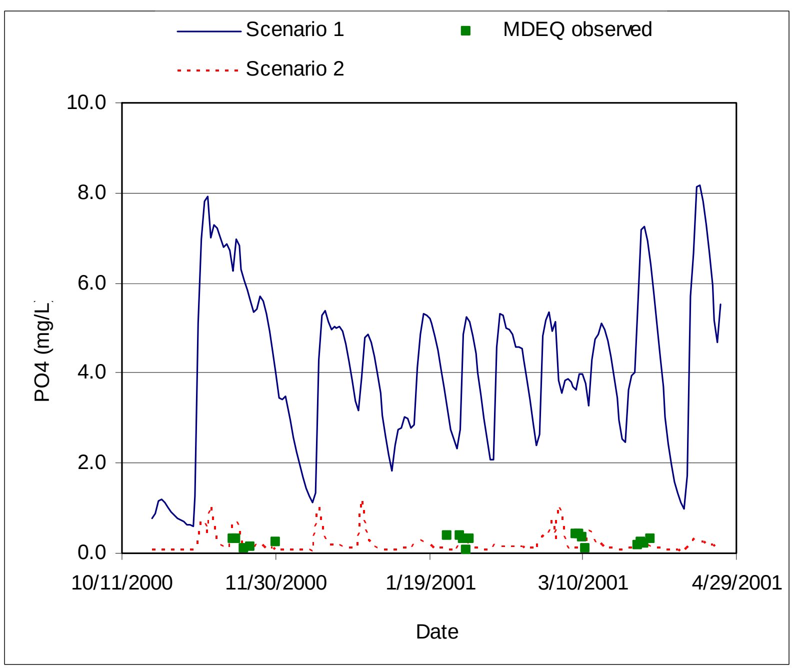


Fig.5.4- 7. Simulated PO_4 at WR2 for modeling scenarios 1 and 2

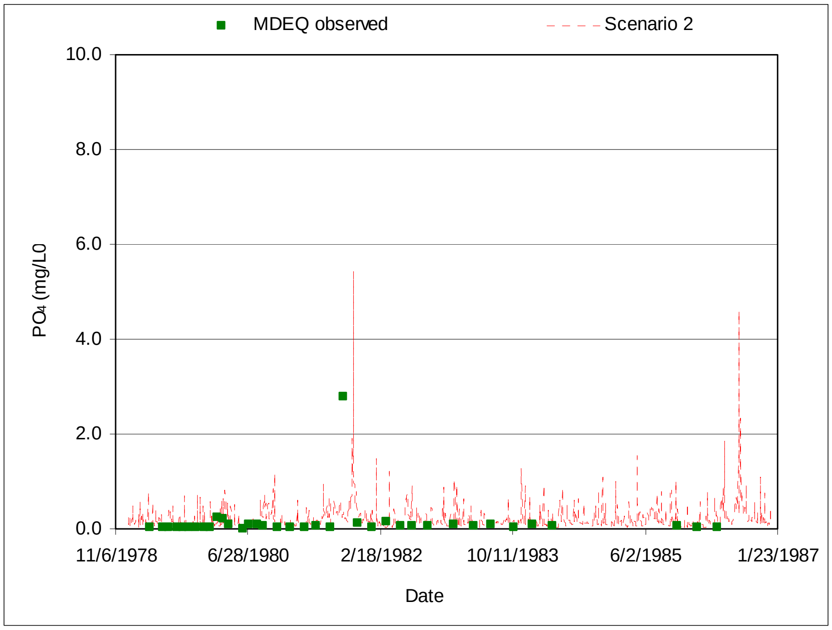


Fig.5.4- 8. Simulated PO_4 at USGS 02481510 for modeling scenario 2

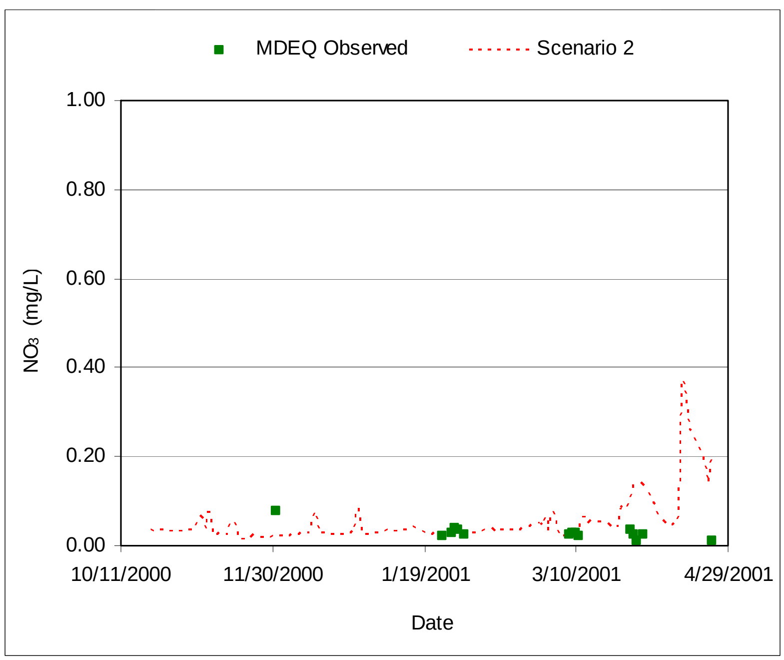


Fig.5.4- 9. Simulated NO_3 at JR3 for modeling scenario 2

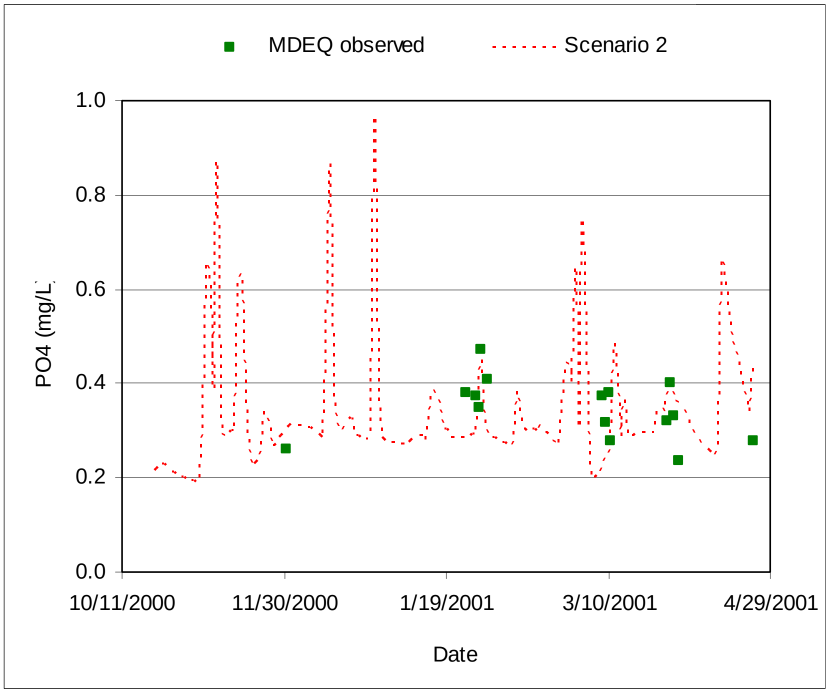


Fig.5.4- 10. Simulated PO_4 at JR3 for modeling scenario 2

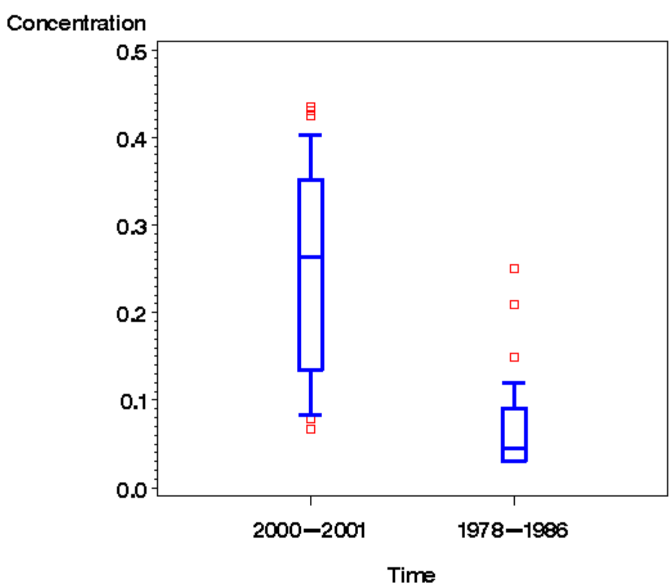


Fig.5.4- 11. Comparison of observed PO_4 between two historical periods: 1978-1986 (USGS) and 2000-2001 (MDEQ).

Contribution Analysis of Nutrient Sources

One of the advantages of mathematical simulation model is that it can help quantify the pollutant contributions from different sources. The nutrient sources simulated included background, point, and non-point sources. The background sources simulated included the direct atmospheric deposition to the stream and Biochemical Oxygen Demand (BOD) decay. The point sources modeled included contributions from National Pollutant Discharge Elimination System (NPDES) permitted point sources, estimated direct discharge from cattle into the stream, and failing septic systems. For the Wolf River watershed, there is one permitted NPDES discharge, located in the northern part. There is no permitted NPDES discharge for Jourdan River watershed in the modeling domain. The non-point sources included atmospheric deposition onto the land surface, fertilization, manure application, and loadings from non-crop land.

Contribution Analysis of NO_3 and NH_4 sources

Fig. 5.5-1 to Fig.5.5-6 displays the results contribution analysis of NO_3 and NH_4 sources. The major contribution of nitrogen was from the point source, and the nitrogen contribution from non-point source is second largest in magnitude (Fig. 5.5-1 to Fig.5.5-6). The magnitude of nitrogen contribution from background sources was negligible compared with point and non-point sources (Fig. 5.5-1 to Fig.5.5-6).

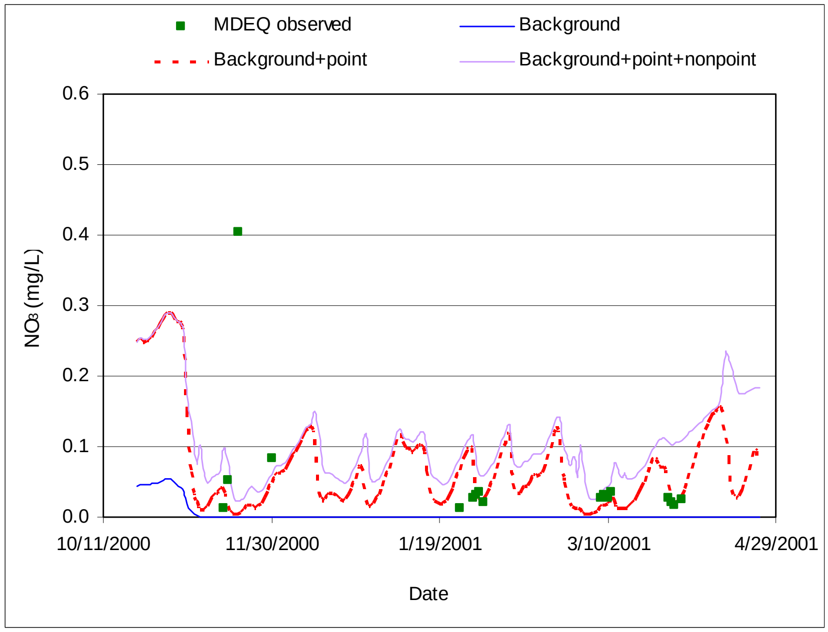


Fig.5.5- 1. Contribution analysis of NO_3 simulation at WR2 for scenario 2

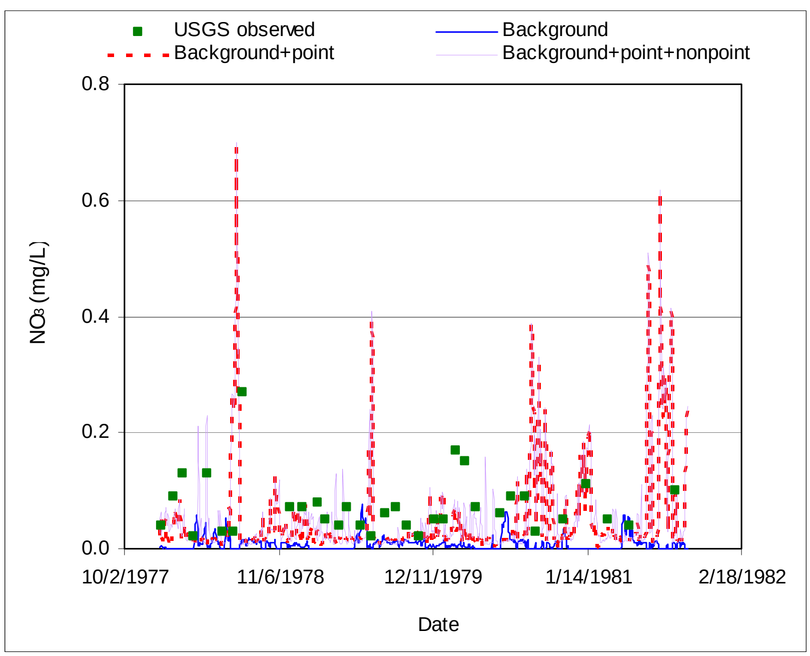


Fig.5.5- 2. Contribution analysis of NO_3 simulation at USGS 02481510 for scenario 2

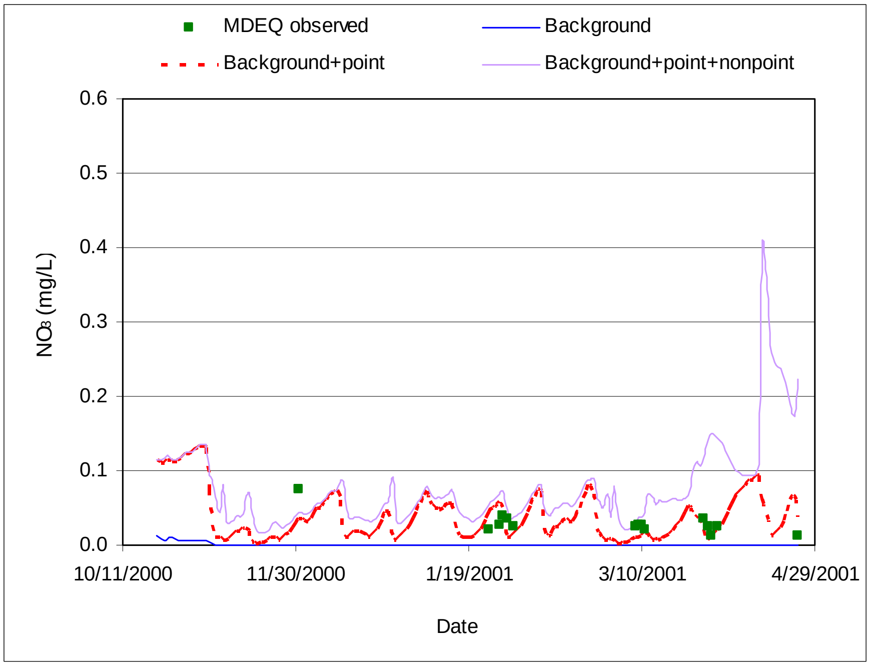


Fig.5.5- 3. Contribution analysis of NO_3 simulation at JR3 for scenario 2

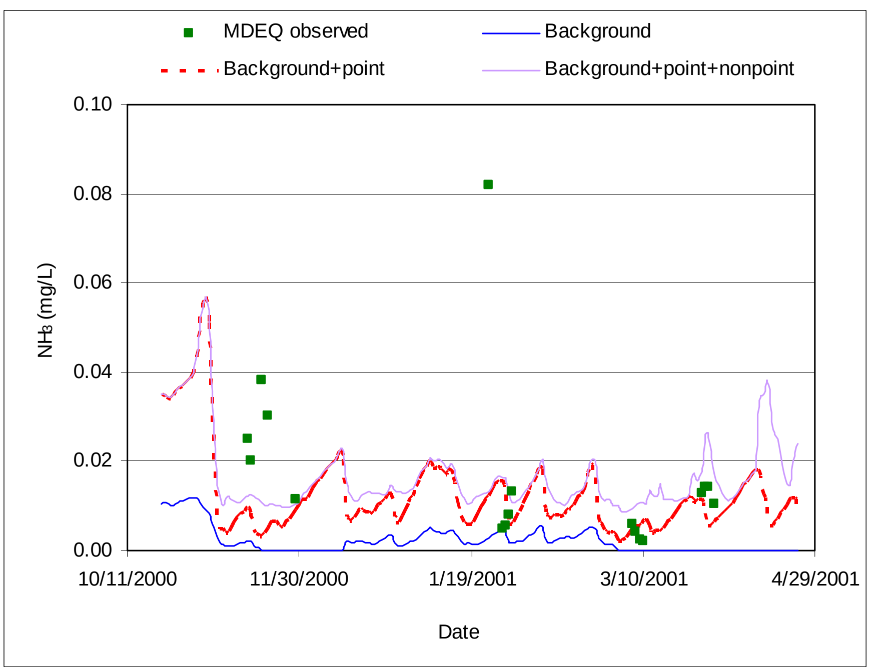


Fig.5.5- 4. Contribution analysis of NH_4 simulation at WR2 for scenario 2

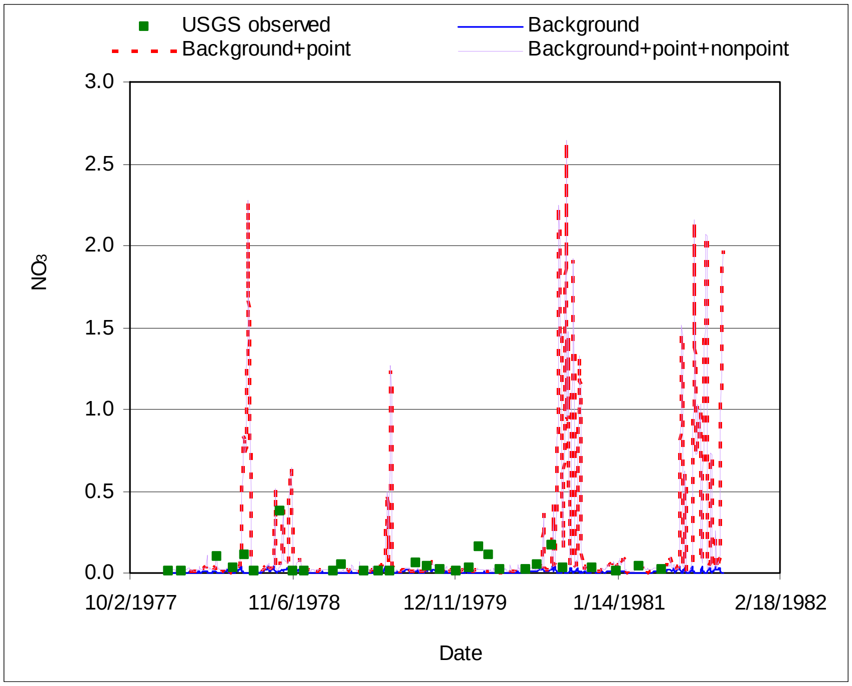


Fig.5.5- 5. Contribution analysis of NH_4 simulation at USGS 02481510 for scenario 2

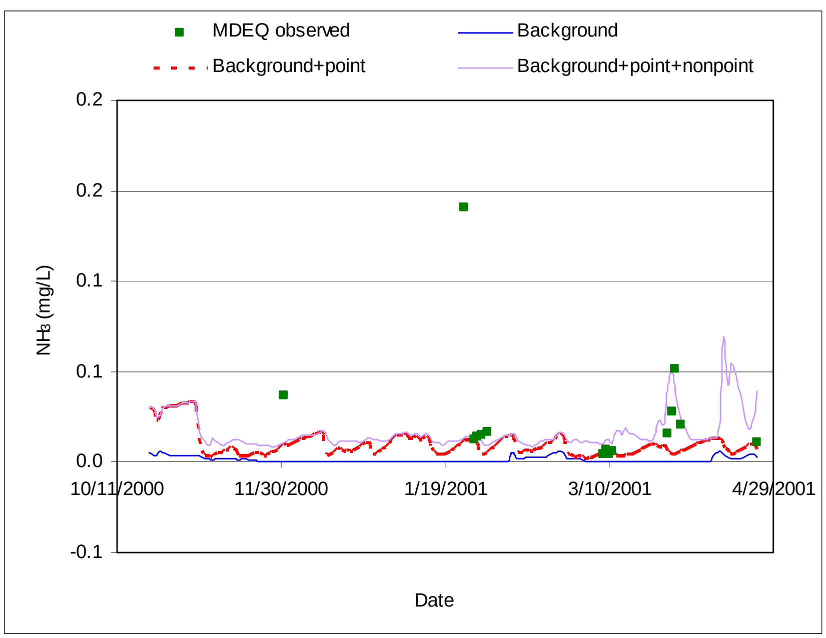


Fig.5.5- 6. Contribution analysis of NH_4 simulation at JR3 for scenario 2

The simulated NH_4 spikes by the developed model were caused by the point-source pollutants (Fig. 5.5-5). In addition, these spikes were all associated the simulated extreme low flow events. Over the time period of 1978 to 1981, there were 23 spikes with simulated NH_4 concentrations higher than 1.4 mg/L (Fig. 5.5-5). The average simulated flows associated with these 23 NH_4 spikes ranged from 13.5 to 26.3 cfs, whereas the average flow over this time period was 967.5 cfs (Fig.5.5-7).

Among the point sources, failing septic system contributed the largest loadings of NH_4 compared with NPDES source and direct contribution from cattle discharge (Fig. 5.5-8). In this model, the contribution from failing septic system was treated as point source and was directly discharged into the stream. Generally, the depth of installed septic system is about 24 inches. Hence, it is most appropriate to link the pollutant loadings from failing septic systems to the lower layer, ranging from 6.5 to 47.5 inches prescribed in the model. The model users can use Monthly Data Block or Manual Time Series method to enter the nutrients into the surface layer (0-0.5 inches) and/or upper layer (0.5-6.5 inches). However, how to input the nutrient into the lower layer and groundwater layer (47.7 to 1335 inches) is unclear. Filoso et al. (2004) entered the nitrogen loadings from failing septic system into the surface layer and pointed out that it was not the best choice. In the developed model, it was also assumed that the daily loadings from failing septic system were constant over the entire simulation period, no matter how much the simulated stream flow is. Hence, the manner of treatment of failing septic system caused the simulated high NH_4 spikes in the extreme low flow events (Fig. 5.5-5). As expected, after excluding the loadings from failing septic systems, the simulated high NH_4 spikes decreased drastically (Fig.5.5-9).

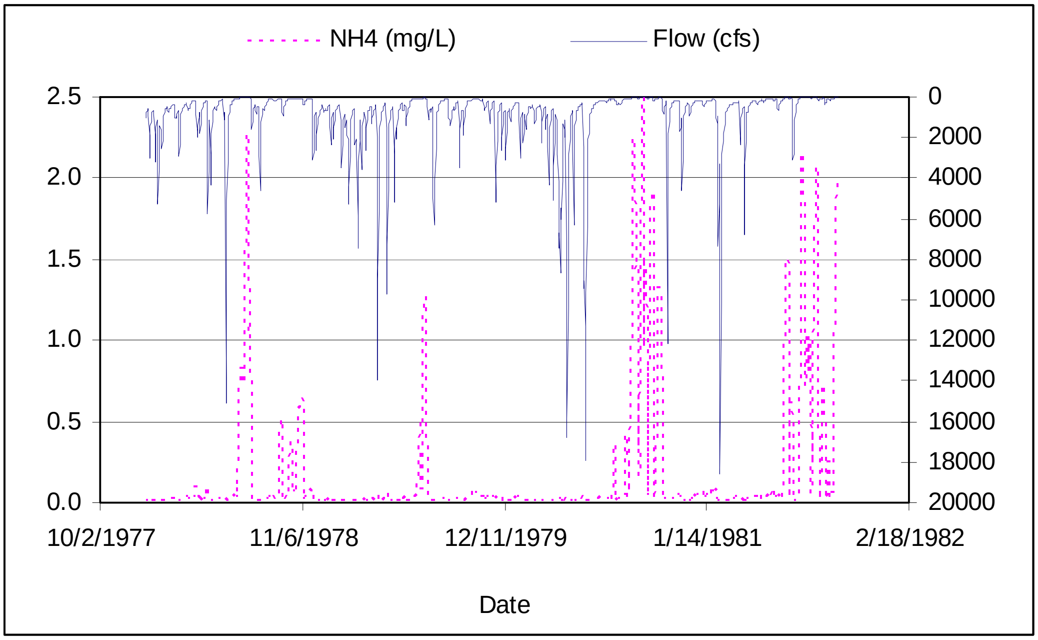


Fig.5.5- 7. Simulated flows vs. NH_4 at USGS 02481570.

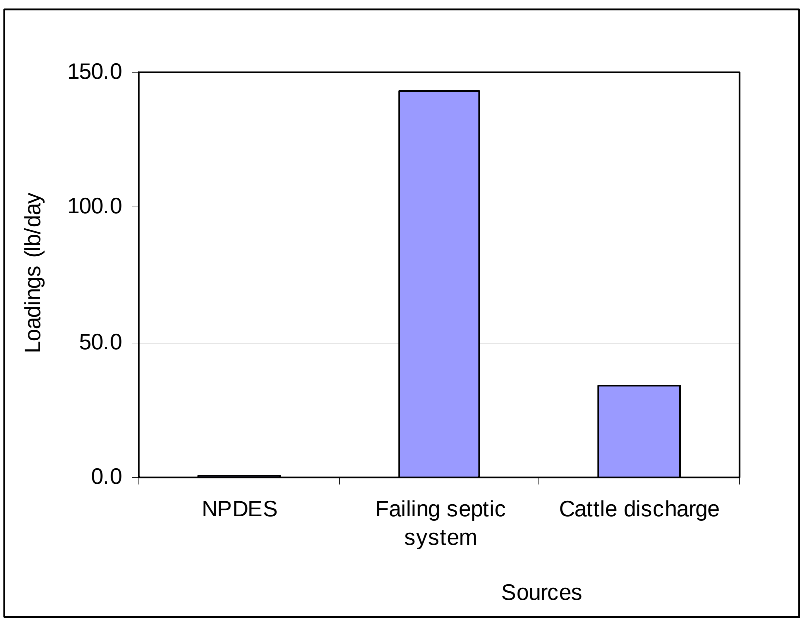


Fig.5.5- 8. NH_4 contributions from different point sources.

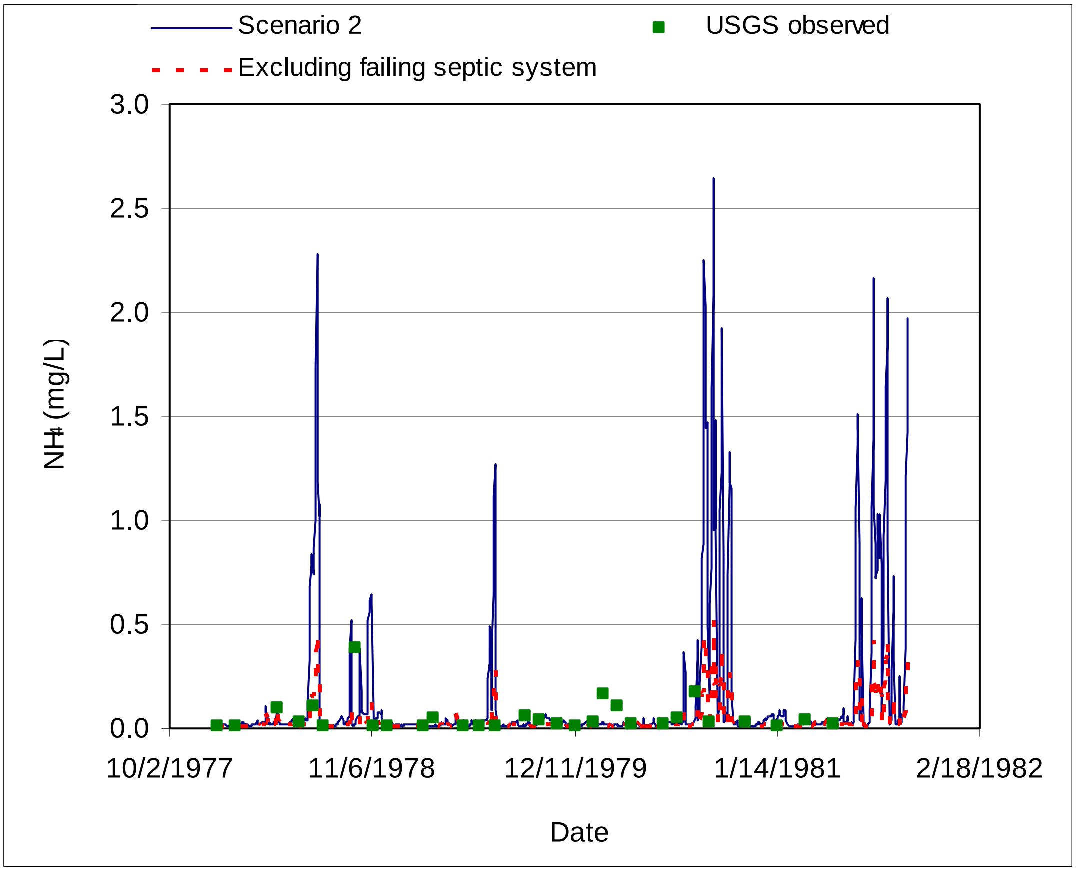


Fig.5.5- 9. Impacts of failing septic system on NH_4 simulation.

Contribution Analysis of PO_4 Sources

Fig.5.5-10 to Fig.5.5-12 displays the results of contribution analysis for PO_4 sources. Contrary to nitrogen simulation, the major contribution of PO_4 was from non-point sources, and the magnitude of contributions of background and point sources were negligible compared with that of non-point sources (Fig.5.5-10 to Fig.5.5-12). The simulated high PO_4 spikes, 5.43 mg/L on October 17, 1981 and 4.6 mg/L on July 30, 1986, were also caused by the treatment of failing septic system as point source (Fig.5.5-13).

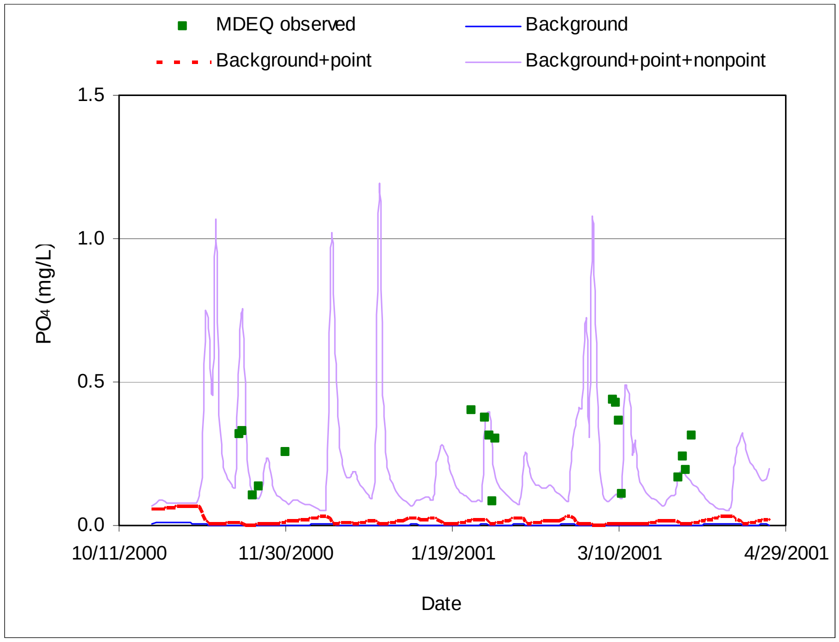


Fig.5.5- 10. Contribution analysis of PO_4 simulation at WR2 for scenario 2

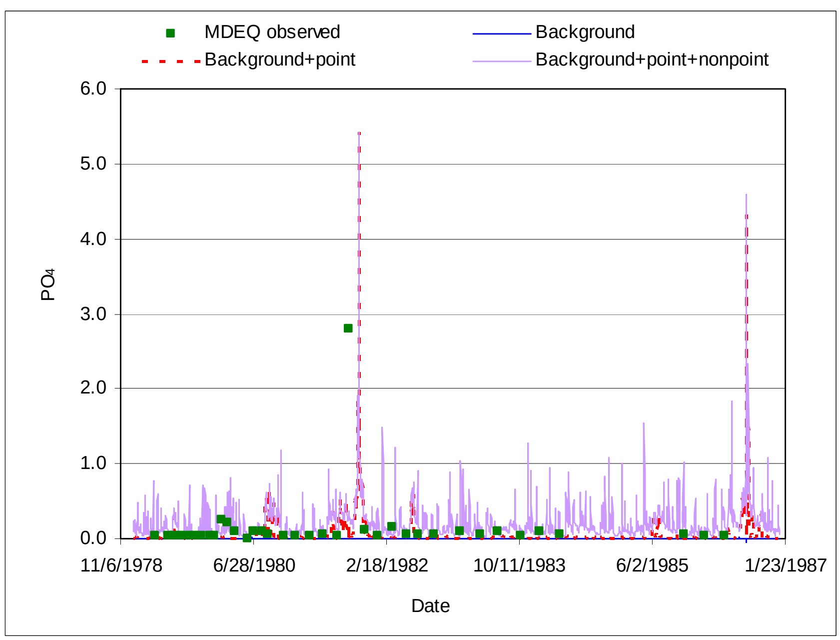


Fig.5.5- 11. Contribution analysis of PO_4 simulation at USGS 02481510 for scenario 2

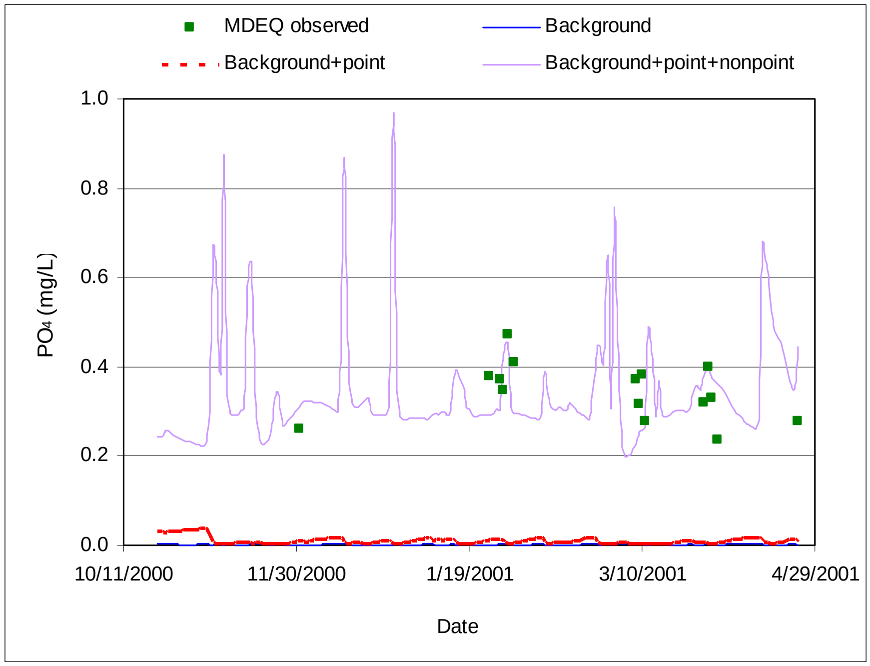


Fig.5.5- 12. Contribution analysis of PO_4 simulation at JR3 for scenario 2

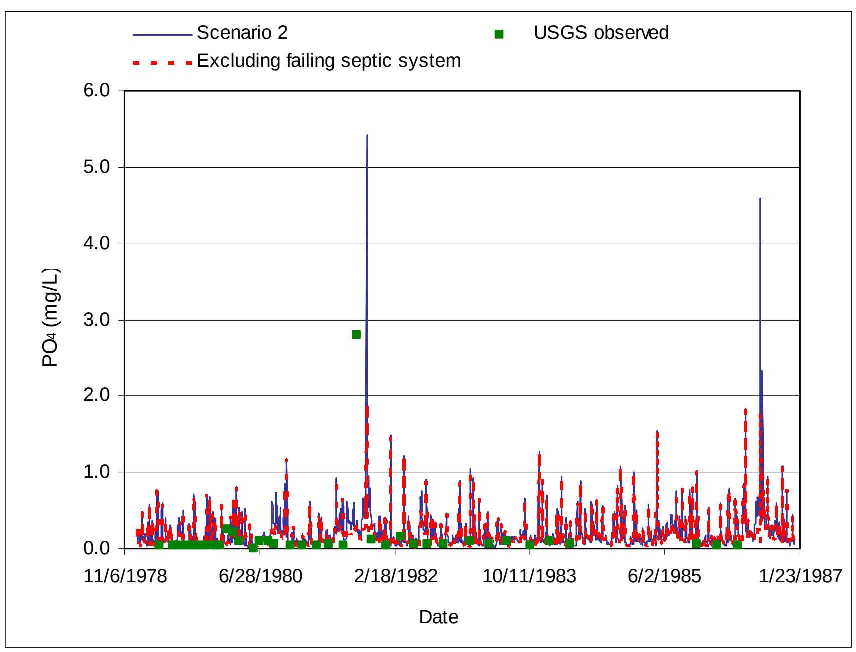


Fig.5.5- 13. Impacts of failing septic system on PO_4 simulation.

CHAPTER VI

SUMMARIZATION

The development of a watershed hydrology and water quality model is very useful for determining the TMDL. The purpose of this study is to calibrate the initially developed water quality model of St. Louis Bay watershed for MDEQ with the purpose of TMDL determination.

Generally, the development of a watershed computational model involve in the following steps: 1) environmental problems identification, 2) model selection, 3) spatial data base development, 4) field data collection and analysis, 5) parameter estimation, and 6) model calibration. For this study, the focuses are on field data analysis, parameter estimation, and model calibration. The development and calibration of a watershed model takes lots of time and efforts, especially for the use of AGCHEM modules. What I have learned from this modeling research can be summarized as follows.

- The model inputs of nutrient distribution in the soil has been proved to be valid based on extensive soil sampling research in both the St. Louis Bay watershed and other watershed with similar characteristics.
- The phosphorus input form for the St. Louis Bay watershed model, was substantiated by the results from edge-of-field experiment. In addition, the validity of phosphorus mass balance applied in the model was also proved by the edge-of-field experiment.

- Whether the modeler should use AGCHEM modules to simulate the nutrient transportation in the cropland depends on the modeling purpose, data availability and watershed characteristics. AGCHEM module has advantages in simulating the complex nutrient source, sink, and transformation processes. However, the use of AGCHEM module requires too much information and extensive data and calibration efforts. Under the condition that the watershed of interest is an agriculture-intensive area and there is enough information to support the model development, the use of AGCHEM is the best choice. However, the use of AGCHEM module with paucity of information introduces uncertainty. For the St. Louis Bay watershed, the water is clean in terms of nitrogen level in the stream. In addition, the cropland covers only 3% of the total area. Hence, the simple method, PQUAL, may be a better choice for simulating the nitrogen transportation. The use of AGCHEM modules for PO4 simulation may be necessary since the phosphorus level is more acute.
- The overall model performance is responsive to the manner in which loads are applied. Hence, more attention should be focused on the correct estimation of boundary loading forcing functions instead of iterative calibration of input parameters. The modeling performance depends on the correct characterization of types, locations, and magnitudes of the pollutants of concern. For the long period modeling, a better choice is to develop the nutrient inputs from the fertilization based on the average loadings for the simulation period instead of the most recent recommended fertilization rates.

- It is very important to make sure that what parameters can be calibrated and what parameters can not. The developed nutrient inputs for the boundary loading functions, such as from atmospheric deposition, fertilization practice, manure application, failing septic system, NPDES source, and failing septic system, should **not** calibrated. The parameter, annual uptake target of nutrients, must be calibrated. The generated amount of annual nutrient uptake strongly depends on soil nutrient and moisture level. For the St. Louis Bay watershed, the annual nitrogen uptake target for corn was calibrated to be 1,745 lb/ac in order to generate the intended plant uptake of 50.7 lb/ac. The input of nitrogen uptake target of 50.7 lb/ac would generate the actual nitrogen uptake of 22.2 lb/ac, which misrepresents the nitrogen mass balance. ACQOP, accumulation rate of pollutant for land segment, is a site-specific parameter. The value of ACQOP needs to be calibrated to match the observed data.

- For the long period modeling, if there are indications of obvious land use changes, separate modeling should be considered. The St. Louis Bay watershed modeling spans a long period, from 1965 to 2001. There is much difference in observed DO and PO_4 between two historical periods: 1978 to 1986 and 2000 to 2001. Especially for the observed PO_4, the median value was 0.06 and 0.26 mg/L, respectively. This change in observed data could be caused by land use change. It is recommended that the impacts of land use change on water quality simulation be studied.

- Contribution analysis of pollutant sources is a very effective method to help calibrate the developed model. During the calibration of the St. Louis Bay watershed model, the contributions of pollutant from background, point, and non-point sources were stepwise-added to examine the modeling performance. This will allow the modelers to compare the contributions from different sources and provide the basis for TMDL determination.

- The spatial analysis of observed water quality data can give an insight of how much sub-watersheds should be delineated. In the sub-watershed 018 of Wolf River, the temporal trends of DO, BOD, NO_3, and NH_4 were consistent among the sampling stations WR2, WR3, WR4, and CRN1, whereas the trend of observed PO_4 was not consistent. This indicated that the current delineation of the Wolf River watershed is appropriate for DO, BOD, NO_3, and NH_4 modeling, but less appropriate for PO_4. Since PO_4 concentration is a primary concern, it is recommended that refinement of sub-watershed 018 should be considered in future model development efforts.

When developing the St. Louis Bay watershed model, many assumptions were taken to develop the input parameters related to boundary loading functions. These assumptions include:

- A typical or representative agriculture practices, including plant, harvest, and fertilization data, were assumed to be taken for the whole watershed, and repeated every year over the entire simulation period.

- For nutrient input from fertilization, 25% of the applied nitrogen was assumed to be in the form of NO_3, and 75% was in the form of NH_4. The phosphorus fertilizer was assumed to 100% in form of PO_4.
- The broadcasted nutrient was assumed to be applied into the surface zone, and the injected was assumed to be applied into the upper zone. For the incorporated nutrient, 10% was assumed to be applied into the surface zone, and the remaining 90% was incorporated into the upper zone.
- The nutrient efficiency of hay, the ratio of amount of nutrient by plant uptake to the amount of nutrient by fertilization, was assumed to be 70%. The fertilizer applied to the hay cropland was assumed to be triple thirteen (13-13-13).
- All manure produced was assumed to be only applied to hay cropland.
- For manure application, 60% of nitrogen was assumed to be ORN, and 40% was in the form of NH_4. The phosphorus was assumed be 50% in ORP and 50% in PO_4.
- For atmospheric deposition, the observed data, at LA30 NADP/NTN site, was available since 1983. The average value of nitrogen from 1983 to 2001 was assumed to be representative the whole simulation period.
- To calculate the nutrient input from dry atmospheric deposition, the ratio of nutrient from dry atmospheric deposition to wet atmospheric deposition was assumed to be 0.7.

- The nutrient contribution from NPDES source, located in Alligator Creek with permit number of MS0031330, was determined based on best professional judgments, and the daily loading rates were assumed to be constant throughout the year.
- For the nutrient contribution from failing septic system, it was assumed that the failing rate of septic system was 50%. The discharge of 70 gallons of wastewater per person was assumed as the average daily load. The phosphorus was assumed to be 62.5% in the form of PO_4 and 37.5 in the form of ORP.
- The nutrient contribution from failing septic systems was treated as point source pollutants, and daily loading rates were assumed to be constant throughout the year.
- For the direct nutrient contributions from cattle, 4% of waste from grazing cattle was assumed to be directly discharged into the stream. The nitrogen was assumed to be 40% in the form of NH_4, and 60% in the form of ORN. The phosphorus was assumed to be equally distributed between PO_4 and ORP.

All these assumptions were made based on best professional judgments and consultations with experts. These assumptions reflect the typical or representative condition. These assumptions might turn out to be wrong, hence bring a lot of uncertainties. This is the best what we can do to develop and calibrate the watershed model based on limited data. I would like to finish the dissertation with my favorite sentence; "…modeling is a process, not an end (Chapra, 2003)."

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APPENDIX A

SUMMARY OF HSPF APPLICATIONS IN JOURNAL PAPERS

Table A- 1 Summary of HSPF applications published in the journal papers.

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size ($mile^2$))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Bicknell, B.R., Donigian, A.S., and Barnwell, T.A. (1984)	Journal of Water Science and Technology	Iowa River Basin, IA, U.S.A.	A typical agricultural watershed with 45% of corn and 22% of soybean cropland Area: 2795.38.	Streamfolow, sediment, pesticide (alachlor), NO_3, and NH_3.	1974 – 1978 Duration: 5 years	The objective is to demonstrate the application of HSPF in a large watershed and evaluate effects of agricultural non-point pollution and BMPs. Extrapolation of calibrated parameters from nearby watersheds was conducted when data were unavailable. Though precipitation data were not representative, modeling performance of flow frequencies ranged from fair to good with deficiencies in simulating snowmelt volume and timing. Poor performance of sediment was attributed to model deficiencies, insufficient calibration, and lack of data. Simulated NO_3 and NH_3 were within the range of observed data. Effects of BMPs were simulated by adjusting soil moisture retention, rainfall interception, surface roughness, and land cover.
Moore, L.W. et al. (1988)	Journal /Water Pollution Control Federation	a small west Tennessee watershed, TN, U.S.A.	An agricultural watershed with 100% corn cropland. Area: 0.07	Runoff, sediment, atrazine, NO_3, NH_3, and TKN	1985-1986 Duration: 19 months	The objective is to simulate the frequencies, quantities, and distributions of pollutants from a small agricultural watershed. The AGCHEM modules were used to simulate nitrogen. The monthly simulation of hydrology and sediments were generally good, whereas simulated monthly NO_3, NH_3 were fair. The long-term simulated atrazine was twice of observed. Modeling performance of all constituents for individual storm events ranged from good to poor.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size ($mile^2$))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Ng, H.Y.H. and Marsalek, J. (1989)	Water Resources Bulletin	Waterford River Basin, Newfoundland, Canada	More than 30% is forested and more than 5% is urbanized. Area: 20.46	Snowmelt and streamflow	1981-1983 Duration: 29 months	The objective is to assess impacts of urban development on watershed hydrology. Monthly simulation of streamflows was very good and the absolute error percentage between simulated and observed was 20%. The urban development could result in increase in peak flows and incidence of flooding, but would not affect the streamflow volumes very much. The peak flow would be increased by 20% under the scenario of doubling the impervious area.
Chew, C.Y., Moore, L.W., and Smith, R.H. (1991)	Research Journal of the Water Pollution Control Federation	North Reelfoot Creek watershed, TN, U.S.A	A typical agricultural watershed with 54.5% of cropland Area: 56.37	Streamflow and sediment	1984-1988 Duration: 54 months Calibration: 1984-1986 Verification: 1987-1988	The objective is to study nonpoint source pollution from agriculture and demonstrate that HSPF is a useful tool for watershed management. The implementations of two BMPs were simulated. Generally, the monthly and annual simulations of streamflow and sediment were good. Modeling performance of streamflow and sediments for individual storm events varied from good to poor.
Moore, L.W. et al., 1992	Water Environment Research	North Reelfoot Creek watershed, TN, U.S.A	A typical agricultural watershed with 54.5% of cropland Area: 56.37	Streamflow and sediment	1984-1988 Duration: 54 months Calibration: 1984-1986 Verification: 1987-1988	The objective is to evaluate impacts of different BMPs on sediment productions. Three BMPs, conversion from cropland to grassland (BMP#1), conservation tillage systems (BMP#2), and reservoir construction (BMP#12), were compared. BMP#1 resulted in the greatest sediment reduction, whereas BMP#2 only achieved moderate reductions. Construction of three reservoirs only achieved 50% of sediment reduction as BMP#1. BMP#1 was the most cost-effective approach.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size ($mile^2$))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Rahman, M. and Salbe, I. (1995)	Environment International	South Creek catchment, Sydney, Australia	An agricultural watershed with 66.7% of pasture and cropland. Area: 139.00	Streamflow, DO\BOD, TN and TP	Not mentioned.	The objective is to quantify the nutrient contributions from point and diffuse sources. The watershed model was successfully calibrated for streamflow and nutrients. Results from frequency duration analysis indicated that contributions from nonpoint sources are significant for a substantial proportion of time and contributions from point sources is quite too high. Unfortunately, the author did not show any calibrated results for streamflow and nutrients.
Laroche, et al. (1996)	Journal of Environmental Engineering	Agricultural Canada experimental farm, Quebec, Canada	An agricultural watershed with 100% of cropland. Area: 0.30	Streamflow and atrazine	1991-1993 Duration: 29 months Calibration: 1991-1992 Verification: 1993	The objective is to assess HPSF performance in simulating pesticide movement in an agricultural watershed. Modeling performances for daily, weekly, and monthly streamflows were quite good, with correlation coefficients all higher than 0.67 for both calibration and confirmation periods. The simulated atrazine was in the same range of observed data.
Tsihrintzis, V.A. et al. (1996)	Water Resources Bulletin	West Wellfield Interim Protection Area, FL, U.S.A.	An urbanized watershed with 40% of urban area Area: 117.19	Streamflow, sediment, and NO_3.	1990 Duration: 1 year	The objective is to assess impacts of agricultural and urban activities on water quality. Reliability of the developed model is questionable since the author did not calibrate the streamflow, which is the prerequisite of water quality modeling. The results of model test runs indicated that application of minimum fertilizers and replacement of fertilizers by sewage sludge resulted in pollution reduction. Conversion from agricultural to urban also caused pollution reduction.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Fontaine, T.A. and Jacomino, V.M.F. (1997)	Journal of the American Water Resources Association	White Oak Creek catchment, TN, U.S.A.	A forested watershed with 80% of forest land, 10% grass land, and 10% developed land. Area: 6.20	Streamflow, sediment, and Cs^{137}	Mar. 1 to June 8, 1990 Duration: 3 months	The objective is to evaluate the usefulness of sensitivity analysis on watershed modeling. The simulated streamflows were less sensitive than sediment and Cs^{137} to changes of input parameters. The results of sensitivity analysis could help with model selection, planning data collection, model calibration, and uncertainty evaluation.
Jacomino, V.M.F. and Fields, D.E. (1997)	Journal of the American Water Resources Association	White Oak Creek catchment, TN, U.S.A.	A forested watershed with 80% of forest land, 10% grass land, and 10% developed land. Area: 6.20	Streamflow, sediment, and Cs^{137}	Mar. 1 to June 8, 1990 Duration: 3 months	The objective is to explore an approach of calibrating a HSPF in a small catchment where few parameters estimates are available. A critical approach was developed by combining sensitivity analysis, numerical optimization, and testing of derived input parameters outside the calibration period to enhance the predictive capability of simulation model.
Chen, Y.D. et al. (1998a and 1998b)	Journal of Environmental Engineering	Upper Grande Ronde watershed, OR, U.S.A	A forested watershed with 75% of forest land. Area: 687.26	Stream temperature	1991-1992 Duration: 2 years Calibration: 1991 Validation: 1992	The objective is to enhance the stream temperature simulation by integrating a computer program, SHADE, which generated solar radiation data by combining geometric relationships among the sun position, stream location and orientation, and riparian shading characteristics. HSPF-SHADE modeling system enabled to relate riparian forest management to stream temperature. The diurnal and daily simulations of water temperature were generally good. Sensitivity analysis of water temperature to heat-balance-related input parameters was conducted to facilitate calibration.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size (mile^2))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Srinivasan, M.S. et al. (1998)	Journal of the American Water Resources Association	Purdy Creek watershed, PA, U.S.A Ariel Creek watershed, PA, U.S.A	Glaciated watersheds with more than 50% of forest land. Area: 8.86 and 15.06, respectively	Snowmelt and streamflow	1992-1993 Duration: 19 months Calibration: Purdy Creek Verification: Ariel Creek	The objective is to evaluate the HSPF modeling performance in a watershed with 75% of fragipan soils. HSPF was cable of simulating the fragipan soils by specifying soil conditions with less lower zone storage and higher lateral flow. The model was calibrated in Purdy creek and verified in Ariel Creek watershed. Model performance is good based on absolute error percentage. However, the model was unable to project the peak flows for extreme snowmelt events.
Bergman, M.J. and Donnangelo, L.J. (2000)	Journal of the American Water Resources Association	Sebastian River basin, comprising of 9 watersheds, FL, U.S.A	Costal watersheds dominated by pine flatwoods. Area: ranging from 8.44 – 108.70	Streamflow	1991-1995 Duration: 47 months	The objective is to provide freshwater discharge boundary condition for the Sebastian River hydrodynamic and salinity model. A set of hydrologic input parameters was developed and calibrated for three gauged stations based on volume error, visual match, and flow duration. Then, they were applied to the entire area to calculate freshwater discharge.
Brun, S.E. and Band, L.E. (2000)	Computers, Environment, and Urban System	Gwynns Falls watershed, MD, U.S.A.	A urbanized watershed with 48.2% of residential area Area: 65.64	Streamflow	Duration: not clear Calibration: 1973, 1981, and 1990 Validation: 1974, 1980, and 1982	The objective is to examine the impacts of urban development on baseflow and stormflow. The relationships between runoff ration and baseflow as a function of percent impervious cover and percent soil saturation were explored. The baseflow declined by about 20% from pre-urbanized times to 1990. There exited a threshold value of 20% for percent impervious cover. Value of percent impervious cover higher than 20% would cause dramatic change in runoff ratio.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Carrubba, L. (2000)	Journal of the American Water Resources Association	White River basin, IN, U.S.A Albemarle-Pamlico River Basin, VA and NC, U.S.A Apalachicola-Chattahoochee-Flint River basin, AL, GA, and FL.	Dominated by agriculture and forest land. No detailed information. Area: 474.00, 733.00, and 620.00, respectively.	Streamflow	1990-1995 Duration: 6 years Calibration: 1990-1992 Validation: 1993-1995	The objective is to evaluate the accuracy of HSPF application in various geographic regions. Values of r^2 between the simulated and observed were 0.75, 0.44, and 0.69 in calibration run and 0.71, 0.69, and 0.64 in validation run for the three watersheds. Nash Sutcliffe coefficients were low ranging from -0.66 to 0.45 in calibration run and from 0.31 to 0.37 in validation run. The model may not be useful in some geographical areas. However, the author did not give any information of precipitation. For so large watersheds, the error in precipitation data could cause the poor performance of hydrologic modeling.
Bledsoe, B.P. and Watson, C.C. (2001)	Journal of the American Water Resources Association	Hylebos creek watershed and Des Moines Creek watershed, WA, U.S.A.	Urbanized watersheds with impervious area of 18% and 37%, respectively. Area: 5.68 and 5.83	Streamflow	Duration: 40 years	The objective is to explore the relationships between impervious percentage, increases in discharge and stream power, and the risk of channel instability in urbanizing watersheds. The 40-year simulation results indicated that flow regime changed drastically due to the increases in impervious area. The estimated two-year recurrence floods were over four times higher than the estimated pre-development condition. At 18 percent impervious area, the estimated frequency of significant scouring events was increased by fivefold. The estimated frequencies of midbank to bankful flows and significant scouring events were dramatically increased at 37 percent impervious condition.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Chun, K.C., et al. (2001)	Environmental Engineering and Policy	Nakdong River basin, South Korea	With forest and field cover of 69.5%. Area: 9,195.80	N, P, DO, and BOD.	1994-1995 Duration: 2 years	The objective is to assess the impacts of proposed management practices on water quality. The reduction of BOD contributions from industrial point source alone can not achieve significant improvement in water quality. Additional reduction of N and P from domestic and non-point sources needs to be made. The inclusion of three domestic wastewater facilities in the model achieved significant reduction in BOD level.
Al-Abed, N.A. and Whiteley, H.R. (2002)	Hydrological Processes	Grand River watershed, Ontario, Canada	An agricultural watershed, with 78% of agricultural land Area: 2689.20	Streamflow	1981-1985 Duration: 6 years.	The objective is to demonstrate the application of a calibration technique combing GIS with automatic calibration. GIS data were used to develop the starting values for LZSN, UZSN, COVER, and INFILT. The relative magnitude ratios of these parameters among sub-watersheds were kept constant during calibration. This technique generated satisfactory modeling results with error of annual discharge ranging from 4 to 16%.
Bergman, M.J., Green, W., and Donnangelo, L.J. (2002)	Journal of the American Water Resources Association	South Prong watershed, FL, U.S.A.	A mixture of residential development, agriculture, and undeveloped land. Area: 56.25	Streamflow, TSS, TP, and TN	1994-1999 Duration: 6 years	The objective is to provide flow and non-point source pollutants boundary conditions to the Indian River Lagoon hydrodynamics and water quality model. PQUAL module was used to simulate TN and TP. Modeling performance for simulated annual stream flows was satisfactory with Nash Sutcliffe coefficients from 0.44 to 0.85. Modeling performance of TSS, TN and TP varied for individual storm events from poor to good.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Engelmann, C.J.K. et al. (2002)	Journal of the American Water Resources Association	Hellbranch Run watershed, OH, U.S.A	Primarily an agricultural watershed under urbanization Area: 39.78	Streamflow and sediment	1993-1995 Duration: 3 years Calibration: 1993 Validation: 1994-1995	The objective is to assess usefulness of BASINS database for watershed modeling. The results of three-year simulation indicated that observed flows were overestimated by 25% using the single station data in BASINS, whereas the flows were only estimated by 2% using area-weighted precipitation data. The value of r^2 for sediment simulation was 0.36.
Lohani, V., Kibler, D.F., and Chanat, J. (2002)	Journal of the American Water Resources Association	Back Creek watershed, VA, U.S.A	A forested watershed. Area: 57.0	Streamflow	1996-1998 Duration: 24 months Calibration: Oct., 1996-Sep., 1997 Validation: Oct., 1997-Sep., 1998	The objective is to integrate HSPF into a Problem Solving Environment (PSE) for simulating alternative watershed management practices. The HSPF-PSE modeling system provided more user-friendly interface and better output display capability. Urbanization affected watershed hydrology in a complex manner. Impacts of three urbanization scenarios were investigated. The percentage of impervious area was consistently significant in the Back Creek watershed.
Bosch, D.J., et al. (2003)	Journal of Water Resources Planning and Management	Back Creek watershed, VA, U.S.A.	No detailed information. Area: 55.98	Streamflow	1995-1998 Duration: 4 years Calibration: 1995-1997 Validation: 1998	The objective is to assess the impacts of residential settlement forms on hydrology and local government costs and revenues. Under the assumption of fixed increase in population, low density development had not only the highest impacts on streamflows for the resulted highest pervious area per person, but also the highest increase in revenues to local government from public sewage, water, and transportation costs.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Doherty, J., and Johnston, J.M. (2003)	Journal of the American Water Resources Association	Four non-overlapping watersheds in the Contentnea Creek basin, NC, U.S.A.	Primary land uses are forest, agriculture, and pasture. Area: 487.38, 156.58, 82.52, and 90.14, respectively.	Streamflow	1970-1995 Duration: 26 years Calibration: 1970-1985 Validation: 1986-1995	The objective is to apply automated Parameter Optimization software (PEST) to facilitate model calibration and predictive analysis. A regularization methodology, which minimized parameter differences between watershed models while keeping the fitness between simulated and observed data, were used to calibrate similar watersheds simultaneously. The extent of model predictive uncertainties was explored by the PEST's nonlinear predictive analysis functionality.
Endrey, T.A., et al. (2003)	Hydrological Processes	Powerstation watershed, NY, U.S.A.	A forested watershed with glacial tills Area: 0. 17	Streamflow	December, 2001 Duration: 1 month	The objective is to evaluate the sensitivity of HSPF hydrograph to three land cover map inputs: GIRAS, MRLC, and DOQQ. The HSPF hydrograph was found to be very sensitive to different land use input data in terms of peak flows ranging from 35% underestimation to 20% overestimation. The differences in model performance were due to different algorithms used in the land use map to estimate impervious area.
Im, S., Branna, K.M., and Mostaghimi, S. (2003)	Journal of the American Water Resources Association	Polecat Creek watershed, VA, U.S.A.	A costal watershed with primary land use of forest. Area: 46.52	Streamflow, sediment, Kjeldal N, NO_3, PO_4, and TP.	1994-2000 Duration: 7 years Calibration: 1994-1995 Validation: 1996-2000	The objective is to assess the impacts of urbanization development on watershed streamflow, sediment, and nutrient loadings. The AGCHEM modules were used to simulate nutrients. Under condition of future urbanization development, the streamflow, peak flow, sediment loads, and NO_3 would increase whereas the loads of total Kjeldal nitrogen, PO_4, and TP would slightly decrease.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size (mile^2))	Modeled constituents	Simulation period	Modeling purpose and conclusions
John, M.S., et al. (2003)	Journal of Hydrology	Part of the Irondequoit Creek basin, NY, U.S.A.	A glaciated undulating watershed with grass/shrub as the main land use Area: 39.19	Streamflow	Aug., 1991-Aug, 1998 Duration: 7 years	The objective is to compare the modeling performances between HSPF and Soil Moisture Routing (SMR). The two models predicted the flows with equal accuracy despite the different runoff mechanisms. HSPF performed a slightly better in predicting winter flows, whereas SMR simulated summer flows a little better. As a distributed model, SMR can capture the spatial variations of soil moisture in a sub-watershed.
Liew, M.W.V., Arnold, J.G., and Garbrecht, J.D. (2003)	Transaction of the ASAE	Little Washita River Experimental watershed, OH, U.S.A.	An agricultural watershed with 66% of rangeland and 18% of cropland. Area: 235.52	Streamflow	1992-2000 Duration: 9 years.	The objective is to evaluate HSPF and SWAT in an agricultural watershed subject to semi-arid climate. Three quantitative and two qualitative criteria were applied. HSPF performed better calibration period, whereas SWAT outperformed in validation. Under much drier condition than average, SWAT exhibited consistent performance, but HSPF gave poor performance. HSPF may provide more accurate prediction of site-specific hydrological response with enough provided information.
Albek, M., Ogutveren, U.B., and Albek, E. (2004)	Journal of Hydrology	Middle Seydi Suyu watershed, Turkey	A forested watershed, but no detailed information. Area: 159.84	Streamflow	1991-1994 Duration: 4 Calibration: 1991-1993 Validation: 1994	The objective is to examine the effects of management practices and climate change. The increase of 3˚C in annual mean temperature would result in decrease of streamflow by 21%. If the deep rooted vegetation covers the entire watershed, simulated streamflow will decrease by 37%, whereas if all vegetation is removed, streamflow will increase by 40%.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Filoso, S. et al. (2004)	Journal of the American Water Resources Association	Ipswich River watershed, MA, U.S.A.	An urbanized watershed with 32% of urban area Area: 155.99	Streamflow, NO_3, and NH_3.	1999-2000 Duration: 2 years	The objective is to predict the impacts of land use on stream water quality. Modeling results indicated stream concentration of NO_3 was four fold as high as prior to urbanization. Conversion of 44% of current forest to urban would result in 30% increase in NO_3 whereas 100% conversion would lead to 100% increase in inorganic nitrogen. Inaccuracies in model prediction were attributed to the treatment of failing septic system and representation of wetland and riparian zone by HSPF.
El-Kaddah, D.N. and Carey, A.E. (2004)	Environmental Geology	Cahaba River watershed, AL, U.S.A.	A forested watershed with 80.27% of forest land. Area: 1,816.01	Streamflow and TN	1989-1992 Duration: 4 years	The objective is to assess water quantity and quality. PQUAL module was used to simulate TN. Simulated streamflows showed good agreement for both low- and high-flow years. However, there was a high difference between simulated and observed TN, which was attributed to the limited point source data and bypass of simulating nitrogen transformation processes.
Hayashi, et al. (2004)	Journal of Environmental Engineering	Yangtze River basin, China	More like of an agricultural watershed with 35% of grassland and 18% of cropland Area: 386,102.16	Streamflow and sediment	1987-1988 Duration: 2 years Calibration: 1987 Verification: 1988	The objective is to test applicability of HSPF in a large watershed using meteorological data from global circulation model (GCM). Generally, the model predicted 5-day average flow well, but underestimated peak flows by up to 71%. Model performance of 5-day sediment was fair. Unsatisfactory performance in flood season was attribute to the GCM precipitation input data, more frequent but less intense than the measured data.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Krause, C.W., et al. (2004)	Journal of the American Water Resources Association	Back Creek watershed, VA, U.S.A.	A mixed watershed with 66% of forest and 17% of urban Area: 57.14	Stream temperature	1996-1998 Duration: 24 months	The objective is to evaluate changes of stream thermal habit under different urbanization level by combining HSPF and Stream Network Temperature Model (SNTEM). Impervious area of 15% would have effects on temperature, but 6% would not have impacts. Flow alteration together with reduced shade and widen channel resulted in an increase of summer water temperature by 1 ˚C. Altered thermal regime could reduce diversity of fish.
Im, S. et al. (2004)	Journal of Environmental Science and Health	Polecat Creek watershed, VA, U.S.A.	An urbanizing watershed Area: 46.52	Streamflow and fecal coliform	1994-2000 Duration: 7 years Calibration: 1996-2000 Validation: 1994-1995	The objective is to assess HSFP for simulating fecal coliform in an urbanizing watershed. Simulated flows showed good agreement with observed data. In the calibration period, more than 42% of observed data were within the maximum-minimum range of simulated data over the 3-day window, whereas the percentage for validation period was 39.5%.
Paul, S. et al. (2004)	Transaction of the ASAE	Salado Creek watershed, TX, U.S.A.	Area: 192.43	Streamflow and fecal coliform (FC)	1990-1996 Duration: 7 years Calibration: 1990-1993 Validation: 1994-1996	The objective is to evaluate the applicability of HSPF in simulating peak FC and analyze the impacts of uncertainties in most sensitive parameters on model prediction. Predicted peak concentrations were most sensitive to five input parameters: the maximum storage of FC, rate of surface runoff that will remove 90% FC, temperature correction coefficient, water temperature, and decay rate. The major portion of the variance in predicted peak concentrations was caused by the variance of maximum storage of FC.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	**Source of publication**	**Watershed (name and location)**	**Watershed characteristics (nature and size ($mile^2$))**	**Modeled constituents**	**Simulation period**	**Modeling purpose and conclusions**
Saleh, A. and Du, B. (2004)	Transaction of the ASAE	Upper North Bosque River watershed	A rural watershed with primary land use of rangeland and forage field Area: 355.60	Streamflow, TSS, NO_3, organic N, PO_4, and particulate P	1994-1999 Duration: 6 years Calibration: 1994-1995 Verification: 1995-1999	The objective is to evaluate HSPF and SWAT. The AGCHEM modules were used to simulated nutrient processes. HSPF performed much better than SWAT in modeling daily and monthly flows and sediments for both calibration and validation periods, whereas SWAT outperformed HSPF in simulating daily and monthly nutrients.
Ackerman, D., Schiff, K.C., and Weisberg, S.B. (2005)	Journal of the American Water Resources Association	Malibu Creek watershed, CA, U.S.A. Ballona Creek watershed, CA, U.S.A.	An undeveloped and a developed watershed, respectively. Area: 110.43 and 130.50	Streamflow	1989-1998 Duration: 10 years Calibration: 1989-1994 Validation: 1995-1998	The objective is to evaluate HSPF performance in urban arid watersheds. Annual simulations of streamflows were satisfactory in both undeveloped and developed watersheds. Daily simulations of steamflows were poor during extended dry weather periods in the developed watershed, which could be attributed to the poor representation of artificially introduced water from human activities. Hourly simulations of streamflows were unable to capture the timing of peak flows.
Chen, C.W., et al. (2005)	Journal of Environmental Engineering	Mica Creek watershed, ID, U.S.A.	A forested watershed. Area: 10.00	Streamflow	1991-1995 Duration: 5 years	The objective is to compare HSPF with Watershed Analysis Risk Management Framework (WARMF). HSPF was an empirical water budget model and the excess water in the HSPF application was caused by the estimated low evapotranspriation by HSPF. WARME was a mechanistic model suitable to simulate forested watersheds. The developed HSPF model was not useful because of the unrealistically values of DEEPFR and liberal adjustment of LZSN in snow melting events.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size ($mile^2$))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Cryer, S.A., et al. (2001)	Environmental Modeling and Assessment	San Joaquin River watershed, CA, U.S.A.	An agricultural-intensive watershed. Area: Not mentioned	Streamflow	1996-1997 Duration: 1 year	The objective is to provide an in-stream hydrological transportation pathway for insecticide movement. PRZM3 was used to calculate the flow, sediment, and insecticide loadings from the sub-watershed to the stream. Mechanistic model AgDrift was used to account for the spray drift of insecticide. Simulated streamflows by the coupled modeling system were of the same magnitude as the observed data. The timing of hydrograph was also reflected fairly well.
Liu, Z. et al. (2005)	International Journal of Civil and Environmental Engineering	Wolf River watershed, MS, U.S.A.	An undeveloped watershed. Area: 379.69	NO_3, NH_3, and PO_4	1965-2001 Duration: 37 years	The objective is to evaluate impacts of fertilization on the in-stream nutrient simulation. AGCHEM modules were used to simulate nutrients. Fertilization practices had strong impacts on the in-stream nutrients simulations; seasonal variations of nutrient concentrations and the occurrence of simulated peak were associated with the fertilization timing. PO4 loadings from the background and point sources were negligible compared with non-point sources.
Said, A., Stevens, D.K., and Sehlke, G. (2005)	Journal of the American Water Resources Association	Big Lost River watershed, ID, U.S.A.	About 70% of the watershed consists of grasslands and forest. Area: 1436.30	streamflow	2000 Duration: 1 year	The objective is to estimate total water balance using the integrated HSPF-MODFLOW modeling system. Precipitation was deemed to be the major source of water recharge. Approximately 48.37 m^3/s of groundwater returned to surface water in form of baseflow. Estimated amount of water loss out of watershed was 10.44 m^3/s.

Table A-1 Summary of HSPF applications published in the journal papers (continued).

Author(s) (year)	Source of publication	Watershed (name and location)	Watershed characteristics (nature and size (mile^2))	Modeled constituents	Simulation period	Modeling purpose and conclusions
Singh, J. et al. (2005)	Journal of the American Water Resources Association	Iroquois River watershed, IL, U.S.A.	An agricultural watershed with 95% of agricultural land Area: 2,149.82	Streamflow	1972-1995 Duration: 34 years Calibration: 1987-1995 Validation: 1972-1986	The objective is to compare modeling performance of HSPF and SWAT in a large agricultural watershed. The application of HSPF required much more efforts than SWAT in preparing climate data. The results of 24-year simulation indicated that the simulated daily, monthly, and annual streamflows were similar. SWAT projected low flow events slightly better than HSPF. The over-estimation of low flow events by HSPF may be caused by the comparatively low potential evapotranspiration input from BASINS database.
Vivoni and Richard, (2005)	Journal of Hydroinformatics	Williams River watershed, New South Wales Australia	A forested watershed. Area:486.49	Streamflow	1988-2000 Duration: 12 Calibration: 1988-1995 Verification: 1996-2000	The objective is to demonstrate the loosely coupling of the GIS-based field data sampling and watershed hydrology and water quality modeling. The GIS technologies were used to select sampling sites, map spatial variations of hydrologic and water quality parameters, and facilitating watershed hydrologic modeling. The hydrologic model was calibrated and verified at Mill Dam Fall and Tillegra with Nash-Sutchliffe coefficient ranging from 0.16 to 0.79.